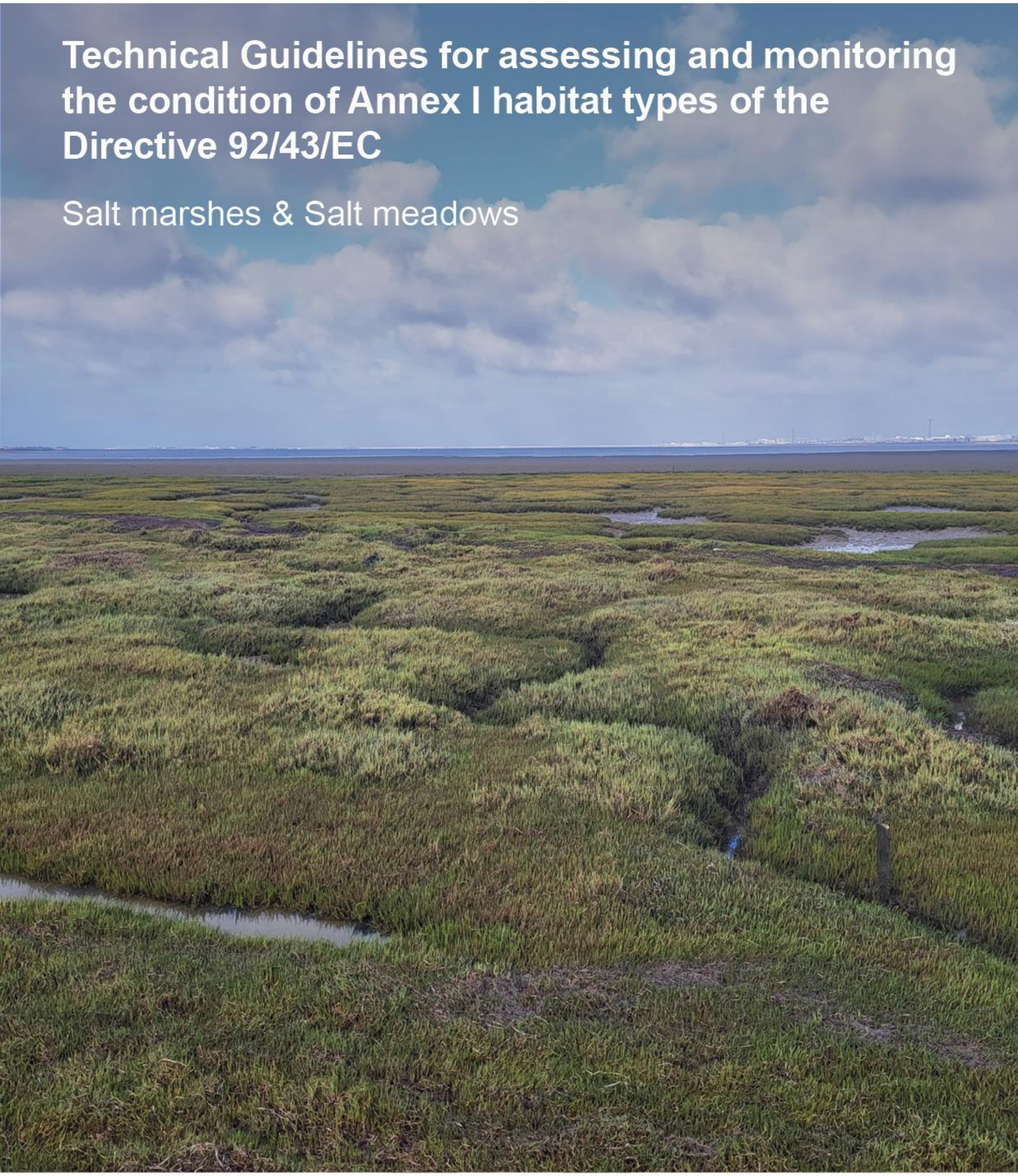
European Commission

Technical Guidelines for assessing and monitoring the condition of Annex I habitat types of the Directive 92/43/EC

Salt marshes & Salt meadows



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Salt marshes and salt meadows

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Contents

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Glossary and definitions

Habitats

Natural habitats: are terrestrial or aquatic areas distinguished by geographic, abiotic and biotic features, whether entirely natural or semi-natural (Habitats Directive).

Habitat condition: is the quality of a natural or semi natural habitat in terms of its abiotic and biotic characteristics. Condition is assessed with respect to the habitat composition, structure and function. In the framework of conservation status assessment, condition corresponds to the parameter “structure and function”. The condition of a habitat asset is interpreted as the ensemble of multiple relevant characteristics, which are measured by sets of variables and indicators that in turn are used to compile the assessments.

Habitat characteristics: are the attributes of the habitat and its major abiotic and biotic components, including structure, processes, and functionality. They can be classified as abiotic (physical, chemical), biotic (compositional structural, functional) and landscape characteristics (based on the Ecosystems Condition Typology defined in the SEEA-EA; United Nations et al., 2021).

Species

Characteristic species: are species that characterise the habitat type, are used to define the habitat, and can include dominant and accompanying species.

Typical species: are species that indicate good condition of the habitat type concerned. Their conservation status is evaluated under the structure and function parameter. Usually, typical species are selected as indicators of good condition and provide complementary information to that provided by other variables that are used to measure compositional, structural and functional characteristics.

Variables

Condition variables: are quantitative metrics describing individual characteristics of a habitat asset. They are related to key characteristics of the habitat that can be measured, must have clear and unambiguous definition, measurement instructions and well-defined measurement units that indicate the quantity or quality they measure. In these guidelines, the following types of condition variables are included:

- **Essential variables:** describe essential characteristics of the habitat that reflect the habitat quality or condition. These variables are selected on the basis of their relevance, validity and reliability and should be assessed in all MSs following equivalent measurement procedures.
- **Recommended variables:** are optional, additional condition variables that may be measured when relevant and possible to gain further insight into the habitat condition, e.g. according to contextual factors; these are complementary to the essential variables, can improve the assessment and help understand or interpret the overall results.
- **Specific variables:** are condition variables that should be measured in some specific habitat types or habitat sub-groups; can thus be considered essential for those habitats, which need to be specified (e.g. salinity for saline grasslands, groundwater level for bog woodlands, etc.).

Descriptive or contextual variables: define environmental characteristics (e.g. climate, topography, lithology) that relate to the ecological requirements of the habitat, are useful to characterise the habitat in a specific location, for defining the relevant thresholds for the condition variables and for interpreting the results of the assessment. These variables, however, are not included in the aggregation of the measured variables to determine the condition of the habitat.

Reference levels and thresholds: are defined for the values of the variables (or ranges) that determine whether the habitat is in good condition or not. They are set considering the distance from the reference condition (good). The value of the reference level is used to re-scale a variable to derive an individual condition indicator.

Condition indicators: are rescaled versions of condition variables. Usually, they are rescaled between a lower level that corresponds to high habitat degradation and an upper level that corresponds to the state of a reference habitat in good condition.

Aggregation: is defined in this document as a rule to integrate and summarise the information obtained from the measured variables at different spatial scales, primarily at the local scale (sampling plot, monitoring station or site).

Abbreviations

EU**:** European Union

HD – Habitats Directive

IAS – Invasive Alien Species

MS: Member State

EU Member States acronyms:

Austria	(AT)	Estonia	(EE)	Italy	(IT)	Portugal	(PT)
Belgium	(BE)	Finland	(FI)	Latvia	(LV)	Romania	(RO)
Bulgaria	(BG)	France	(FR)	Lithuania	(LT)	Slovakia	(SK)
Croatia	(HR)	Germany	(DE)	Luxembourg	(LU)	Slovenia	(SI)
Cyprus	(CY)	Greece	(EL)	Malta	(MT)	Spain	(ES)
Czechia	(CZ)	Hungary	(HU)	Netherlands	(NL)	Sweden	(SE)
Denmark	(DK)	Ireland	(IE)	Poland	(PL)		

MSFD: Marine Strategy Framework Directive

SEEA EA – System of Environmental Economic Accounting- Ecosystem Accounting

WFD: Water Framework Directive

Executive summary

This technical guidance document presents a common framework for the assessing and monitoring the condition of saltmarshes and salt meadows (habitat types 1310, 1320, and 1330 as listed in Annex I of the Habitats Directive 92/43/EEC). It is part of the initiative led by the European Commission (EC) and the European Environment Agency (EEA) to ensure harmonised, science-based and policy-relevant habitat assessments across EU Member States.

The main objective of this guidance is to support Member States in applying a standardized and consistent approach for evaluating and reporting habitat condition in accordance with Article 17 of the Habitats Directive. The document defines essential, recommended and specific variables for assessment and monitoring, outlines different approaches for setting reference values and threshold to determine good condition and describes methods for the aggregation of results. This framework aims to ensure comparability of assessments across regions and enhance the implementation of national monitoring programmes, which are crucial to inform conservation actions and policy decisions.

Saltmarshes and salt meadows are dynamic habitats of high ecological value, occurring not only in macrotidal coastal zones but also in microtidal systems and inland saline areas. Characterized by halophytic vegetation and influenced by hydrological and salinity regimes, these habitats deliver key ecosystem services such as shoreline stabilization, carbon sequestration, nutrient cycling, and biodiversity support. They are highly sensitive to hydrological alteration, eutrophication, land reclamation, and climate change, underscoring the importance of robust monitoring and targeted conservation.

The concept of Condition Variables is central to this guidance. These are quantitative indicators that reflect habitat quality and functioning, and are classified as essential, specific and recommended variables, depending on their ecological importance and site specificity. Descriptive or contextual variables (e.g. topography, climate, lithology) are also proposed to help interpret results and define ecologically meaningful thresholds, though they are not used directly in condition scoring. Condition variables are assessed to determine the degree of habitat condition (good/nor good) relative to defined thresholds. The aggregation of results from measurement of multiple variables gives a synthetic indicator that provides an integrative view of habitat quality. This approach is used to assess the condition at local scales and further aggregation is then obtains in the corresponding biogeographical applying agreed rules.

The guidance acknowledges key implementation challenges, such as limited data availability, diverse ecological conditions across biogeographic regions, and variability in existing monitoring protocols. While promoting harmonization, the document emphasizes the need for flexibility and adaptation to national contexts. It encourages gradual refinement of variables and thresholds as new scientific evidence becomes available.

To support effective implementation, the document recommends the adoption of standardized methodologies, clear documentation of procedures, and investment in training and capacity-building. It also highlights the potential of emerging tools such as remote sensing and citizen science to enhance data collection and spatial analysis.

By establishing a common methodology for assessing the condition of saltmarshes and salt meadows, this document aims to improve the reliability and comparability of condition assessments across the EU, support biodiversity policy objectives, and contribute to the adaptive management and long-term conservation of these valuable ecosystems.

1. Definition and ecological characterisation

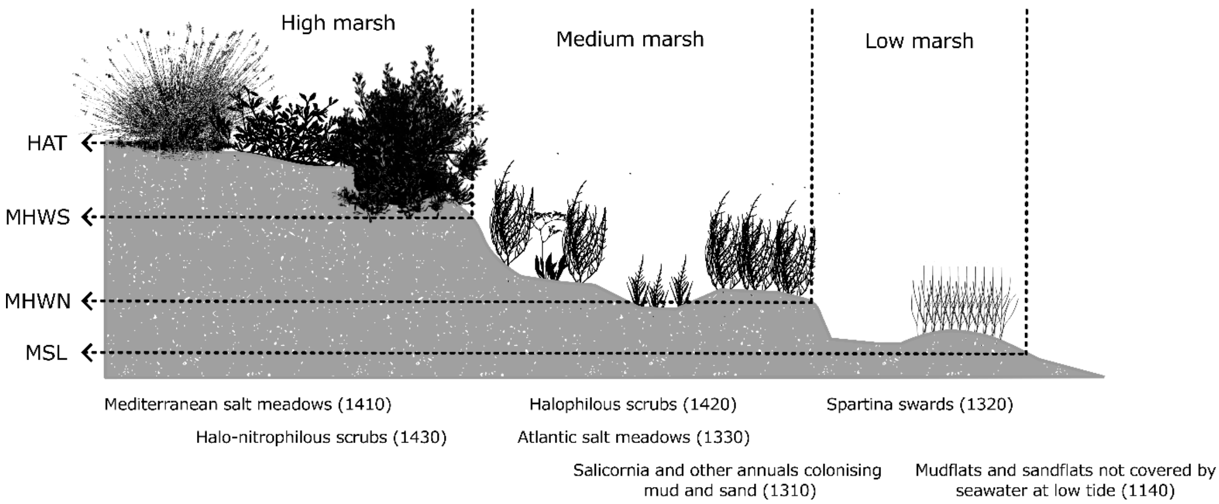
1.1 Definition and interpretation of habitats covered

Salt marshes are coastal wetlands that are periodically inundated in meso- and macrotidal areas, or – at minimum – subject to marine influence in microtidal areas. They are characterised by salt-tolerant vegetation with adaptations to substrate anoxia and desiccation. Salt marshes typically develop in estuarine environments, including coastal bays and barrier beach systems (Adam, 1990; Keith et al., 2020) where natural sediment accumulation is enhanced by the presence of vegetation. As a result, the soil often consists of a variable mix of clay, silt, and fine sand.

Other habitats dominated by halophilous vegetation include non-coastal salt meadows. These occur in natural inland salt basins, in areas with seepage of saline water, or around running or stagnant saline water. Like coastal marshes, these habitats are dominated by salt-tolerant vegetation and often feature reed beds at the edge of brackish waters (European Commission, 2013).

In addition, halophytic habitats also encompass perennial vegetation – primarily scrub –that colonises marine saline muds or grows on dry soils in arid climates.

Figure 1. Distribution of salt marsh habitats along a typical Atlantic coastal profile in Southern Europe



HAT – Highest Astronomical Tide; MHWS – Mean High Water Spring; MHWN – Mean High Water Neap; MSL – Mean Sea Level.

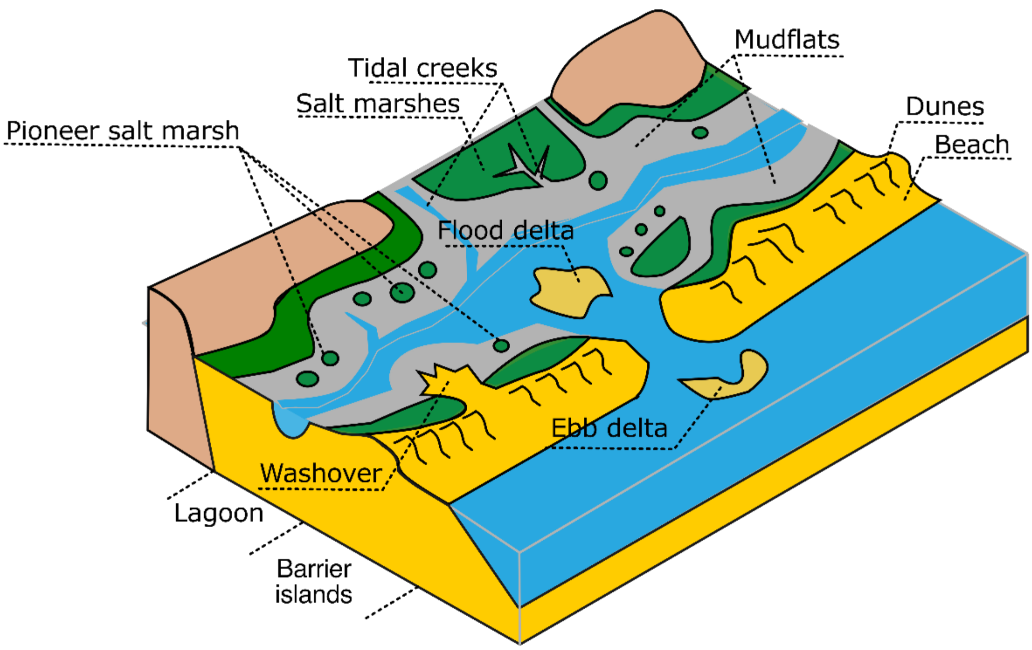
Source: Own elaboration

Salt marshes are widespread habitats along the coasts of Europe. The Atlantic and North Sea coasts host the largest areas of tidal salt marshes (meso- and macrotidal), particularly in the estuaries of Great Britain and the Wadden Sea. In contrast, Atlantic coastal habitats above MHWS elevations and Mediterranean coasts support salt marshes that, although less influenced by tidal dynamics (microtidal in the Mediterranean), are still predominantly shaped by marine processes.

Salt marsh vegetation diversity is highest in these Atlantic salt marshes and in the Mediterranean (Davidson, 2018). Tidal marshes typically host several halophytic habitats (Figure 1) and are always adjacent to habitat 1140. The connectivity between halophytic habitats and adjacent mudflats is crucial, as it can determine the long-term dynamics of the marsh (Bouma et al., 2016). In general, salt marshes and mudflats are subsystems within estuarine environments (Day et al., 2013; Figure 2). Therefore, the guidance provided for assessing the condition of these habitats should be applied complementarily.

Salt marshes provide important ecosystem services, including coastal protection and blue carbon sequestration (De Los Santos et al., 2023; Gilbertson et al., 2020). They are also essential for sustaining coastal fisheries and for the breeding and wintering of migratory birds (Davidson, 2018).

Figure 2. Schematic aerial view of an estuarine system developed landward of a coastal barrier, illustrating its main subsystems



Source: Own elaboration

According to the Interpretation Manual of European Union Habitats (European Commission, 2013), salt marshes and salt meadows are included under subgroups 13 and 14 of the Coastal and Halophytic Habitats. These subgroups comprise salt marshes and salt meadows of the Atlantic (13) and Mediterranean (14) biogeographical regions. All of these habitats are characterised by dominant halophytes – mainly species of Chenopodiaceae, certain herbaceous species, and scrubs – growing in saline soils subject to inundations from various sources, at differing frequencies and durations.

Differences in inundation patterns are especially significant between Atlantic (13) and Mediterranean (14) subgroups, but also between coastal and non-coastal habitat types. Variations in salinity source, inundation period, and frequency influence the dominant vegetation composition, as well as the structure and function of the habitat.

These guidelines propose grouping the different habitat types within subgroups 13 and 14 based on differences in inundation patterns and sources of salinity, according to the following classification.

- **Tidal salt marsh habitats**. This group comprises habitats from subgroups 13 and 14 that are subject to tidal influences. It includes habitat 1310 (*Salicornia* and other annuals colonising mud and sand), 1320 (*Spartina* swards (*Spartinion maritimae*)), and 1420 (Mediterranean thermos-Atlantic halophilous scrub (*Sarcocornetea fruticosi*)). These habitats are typically distributed in bands parallel to Mean Sea Level (MSL), with 1310 and 1320 commonly found in the pioneer and low marsh zones, while habitat 1420 generally occurs in the middle marsh. In these cases, the primary source of salinity is the tidal regime, which follows a regular pattern of inundation frequency and duration. Both are determined by the elevation of the habitat relative to MSL and the local tidal range. Habitats 1310 and 1320 usually occur between MSL and Mean High Water Neap (MHWN), whereas 1420 is found in intertidal areas above MHWN (Figure 1). Habitat 1310 can also occur at higher elevations in topographic depressions (Muñoz-Rodríguez et al., 2017). In microtidal environments, the presence of habitat 1320 maybe be reduced.
- **Coastal salt marshes**. This group comprises coastal habitats from subgroups 13 and 14 located above the Mean High Water Spring (MHWS) level. It includes coastal examples of habitat 1330 (Atlantic salt meadows (*Glauco-Puccinellietalia maritimae*)), 1410 (Mediterranean salt meadows (*Juncetalia maritimi*)), and 1430 (Halo-nitrophilous scrub (*Pegano-Salsoletea*)). These habitats typically correspond to the high marsh zonation (Figure 1; Curcio et al., 2023) and generally exhibit lower soil salinity than habitats in the tidal category. This is due to the less frequent inundation by seawater and the presence of freshwater inputs, such as surface runoff, rainfall, or groundwater seepage, which reduce salinity levels and influence plant community composition.
- **Non-coastal salt meadows**. This category includes examples of habitats 1310, 1340, 1410, 1420, and 1430 occurring far from tidal influences. Most are associated with permanent or temporary lagoons or ponds with brackish water and muddy soils. In these cases, soil salinity typically results from gypsum-rich substrates or saline springs. In non-coastal salt meadows, the frequency and duration of inundation are less regular than in tidal environments and are usually driven by meteorological events (e.g., rainfall versus draught). Vegetation typical of habitat 1310 often occupies silty soils along the retreating waterline. In more permanently inundated zones, brackish-water reed beds characteristic of habitats 1340 and 1410 may be found. In areas subject to strong summer drought, habitats 1420 and 1430 are more commonly established.

Table 1. Categories of salt marsh and salt meadow habitats and their correspondence with Habitats Directive categories

Category/Habitat	1310	1320	1330	1340	1410	1420	1430	Num. habitat types
Tidal salt marshes	X	X	X			X		4
Coastal salt marshes	X		X		X		X	3
Non-coastal salt meadows	X			X	X	X	X	5

1.2 Environmental and ecological characterization and selection of variables to measure habitat condition

Salt marshes and salt meadows are habitats characterised by the dominance of halophytes (Chenopodiaceae, herbaceous plants, and/or shrubs) and by saline, anoxic soils subject to flooding from various sources, with differing frequencies, and durations (Veldhuis et al., 2019). Soil salinity and inundation impose significant physiological stress, and the variation in their sources, frequency, and duration creates environmental stress gradients. These gradients influence: (1) the types of vegetation that can colonise these environments, and (2) the occurrence of spatial zonation patterns in response to environmental gradients.

Accordingly, environmental characterisation should include abiotic variables that explain salinity sources, and inundation frequency and duration. Ecological characterisation should encompass variables that describe vegetation species composition and the spatial patterns of this composition (e.g., zonation). These variables can be grouped into three categories: abiotic, biotic, and landscape.

1.2.1 Abiotic characteristics

The primary abiotic variables for characterising salt marshes and salt meadows can be divided into (a) physical and (b) chemical categories (Table 2). Key physical variables include inundation frequency and duration, as well as sedimentation and erosion dynamics, which can influence inundation patterns by altering the habitat's elevation relative to the source of inundation.

Accurate characterisation of inundation patterns requires an understanding of the inundation source (e.g., tide, river, rainfall, or groundwater), the frequency of inundation, and the elevation of the habitat relative to the corresponding inundation regime.

In tidal and coastal habitats, the inundation pattern is mainly driven by the tidal regime and can be described using the habitat's elevation relative to the lowest astronomical tide (LAT), tidal range, and the incidence of storm surges. In estuaries, riverine water patterns are also important in shaping the inundation dynamics. In all cases, the influence of groundwater and precipitation may be relevant.

For non-coastal habitats, the main variables include rainfall, groundwater level, and/or the presence of groundwater springs. Therefore, the topographic position of the habitat in relation to the groundwater spring level may serve as a proxy for the inundation pattern.

To assess habitat condition, it is important to monitor not only the current state of these physical variables but also long-term threats associated with climate change, such as sea level rise, increased storm frequency, and droughts. Since small changes in habitat conditions can be symptoms of a progressive shift in the stable state due to long-term climate change, and therefore indicate a transition in habitat type.

For salt marshes and salt meadows, the main chemical variables influencing habitat condition include salinity, nutrient levels, and organic matter content (Table 2). Eutrophication represents a significant chemical pressure and threat. Nutrient conditions refer to the availability of carbon, nitrogen, and phosphorus in the soil and groundwater, as well as the presence of other elements such as sulphur (abundant in marine systems as SO_4), potassium (K), sodium (Na), iron (Fe), and calcium (Ca). Evaluating salinity involves measuring salinity levels not only in surface waters but also in soil and groundwater, depending on the source of salinity in the system. Organic matter content in salt marsh soils serves as a key indicator of both soil carbon stocks and the potential development of anoxic conditions.

1.2.2 Biotic characteristics

The presence of habitats from groups 13 and 14 is determined by the dominance of halophytic species. Vegetation in these habitats acts as an ecosystem engineer, and its characterisation is therefore essential for understanding the structure and functioning of the system (Bouma et al., 2005). Biotic characterisation of salt marsh and salt meadow habitats includes compositional, structural, and functional components (Table 2).

Compositional characteristics can be assessed through variables related to plant species assemblages, including both characteristic species and harmful invasive species (HIS). Although species richness is not usually very high in these stress-prone habitats, it remains an important variable, as observed increases or decreases over time may indicate changes in habitat condition.

In salt marshes and salt meadows, key plant species represent a relatively limited group compared to terrestrial habitats. Halophytes are species adapted to varying levels of salinity, anoxia, and inundation (Redelstein et al., 2018). Moreover, even within the same habitat type, dominant species may vary depending on the inundation regime (e.g., intertidal-coastal versus non-coastal habitats such as habitat 1410).

©: Gloria Peralta
Low marsh of *Spartina maritima* with macroalgae present at the Cádiz Bay Natural Park

Harmful invasive species (HIS) include invasive alien species (IAS) or any other invasive species capable of displacing characteristic species from the system. For example, in intertidal systems, *Spartina densiflora* poses a major threat to meridional Atlantic salt marshes. In the Gulf of Cadiz, *S. densiflora* is displacing local populations of *S. maritima*, with an additional risk of viable hybridisation (Curado et al., 2020; Infante-Izquierdo et al., 2021). In the same region, *Spartina patens*, a species native to the Atlantic and Gulf coasts of North America, is also acting as an invasive species, colonising high and brackish marshes (Castillo et al., 2017).

The structural characterisation of salt marshes and salt meadows can be assessed through species distribution and coverage, canopy height, and the presence of zonation patterns. In these habitats, zonation refers to spatial distribution patterns shaped by species' tolerance ranges and physicochemical gradients. These patterns typically appear as parallel bands relative to the waterline in intertidal and coastal areas, or as ring-shaped formations around wet areas in non-coastal habitats. These structural characterisation aims to evaluate the performance of key processes that support habitat persistence. These processes ensure the flow of matter and energy, and help maintain salinity and inundation patterns; otherwise, the system may shift to alternative habitat types. Examples of such processes include productivity and the import or export of organic matter. However, these processes are difficult to assess directly. To date, most Member States evaluate the naturalness of the system as a proxy for its functional integrity.

1.2.3 Landscape variables

To accurately characterise abiotic and biotic factors, a landscape-level evaluation is essential for understanding both local and regional processes. For salt marshes and salt meadows, this evaluation should consider spatial distribution aspects such as extent, patchiness, fragmentation, connectivity to adjacent habitats, and the availability of space for habitat migration as a proxy for resilience. Habitat extent is particularly important for assessing the risk of local extinction; larger habitats generally show greater resilience than smaller ones, especially in salt marshes, where long-term cycles of retreat and expansion are well-documented (Bouma et al., 2016).

©: Gloria Peralta
Medium marsh of *Salicornia* sp. and *Sarcocornia* sp. at the Cádiz Bay Natural Park

In intertidal and coastal habitats, the presence of accommodation space and the absence of human infrastructure are key indicators of resilience, as they provide the conditions necessary for these ecosystems to migrate upland in response to sea level rise (Fagherazzi et al., 2020). Additionally, connectivity features such as natural drainage systems that facilitate water movement are essential landscape and seascape variables for maintaining ecosystem health.

Table 2. Framework for the ecological characterisation and selection of variables for assessing the habitat condition of salt marshes (intertidal and coastal habitats) and salt meadows (non-coastal habitats)

Ecological characteristics	Type	Description	Example variables	
			Intertidal and coastal habitats	Non-coastal habitats
Abiotic characteristics	**Physical characteristics**	Variables that assess inundation patterns	Elevation relative to mean sea level (MSL) Tidal Range Storm surge incidence Rainfall	Spring ground water level Rainfall
		Variables that assess sediment dynamics	Sedimentation and erosion patterns	
	Chemical characteristics	Variables that assess salinity and nutrient conditions	Salinity (groundwater, soil and surface water) Nutrient concentrations (N, P, SO_4, Na, K, Ca, Mg, etc.)	
Biotic characteristics	**Compositional characteristics**	Variables that assess species assemblages	Characteristic species Presence and extent of harmful invasive species (HIS) Species richness	
	Structural characteristics	Variables that describe community structure	Species distribution and coverage Canopy height Zonation patterns	
	Functional characteristics	Variables that assess processes supporting system persistence	Naturalness of hydrological connectivity	
Landscape characteristics		Variables describing habitat extent, spatial scale, and connectivity	Habitat extent Fragmentation Slope	
		Variables describing accommodation space for habitat migration	Area behind the habitat with suitable conditions for upland migration of vegetation	

1.3 Selection of typical species for condition assessment

For a habitat type to be considered at favourable conservation status, the Habitats Directive requires that both its structure and functions are favourable, and that its typical species are also in favourable conservation status (Article 1I).

Typical species are used to assess whether a habitat is in a favourable conservation status (FCS). According to the guidelines for reporting under Article 17 (European Commission, 2023), the selection of typical species for assessing habitat condition should include species that are good indicators of favourable habitat quality and/or species that are sensitive to changes in habitat condition (so-called ‘early warning indicator species’). In addition, they should provide useful supplementary information, assuming that characteristic species are already being monitored as part of the assessment of the habitat’s composition, structure, and function.

Given the ecological and geographical variability of Annex I habitat types, different typical species may occur in different parts of the range of a habitat type or in different subtypes, even within a single Member State or a biogeographical region. However, the total number of sites and occurrences of each habitat type should support viable populations of typical species on a long-term basis within the assessed region, in order for the habitat's structure and functions to be considered favourable. Furthermore, some species may be typical for several habitat types and not dependent on a single Annex I habitat type (European Commission, 2023).

For salt marshes and salt meadows, the most indicative physiochemical characteristics for condition assessment are the inundation pattern, organic matter content, and salinity, although nutrient availability is also relevant. Inundation and salinity cause natural physiological and physical stress on vegetation. As a result, the number of species capable of adapting to these conditions tends to be limited. The combination of inundation pattern, organic matter content, and salinity may define redox conditions and determine which electron donors and microbial pathways predominate (Luo et al., 2019).

The species composition used for condition assessment of habitats of subgroups 13 and 14 may vary depending on the specific characteristics of the habitat, and typical species may differ between sites. Evaluating habitat condition at specific sites therefore requires the verification of locally typical species. In most cases, a decrease in the abundance or distribution of typical species – alongside the occurrence or increase of typical species from adjacent habitats – may indicate changes in abiotic habitat quality (inundation, salinity or nutrient conditions), often as part of the ecological succession following habitat degradation.

The species selected to monitor the condition of this habitat type may vary depending on the specific characteristics of each salt marsh or salt meadow. Although most recorded species are typically vascular plants, representative species may belong to any taxonomic group. The inclusion of lichens, mosses, fungi, and animals – including birds – should also be considered. In salt marshes and salt meadows in particular, animal species can be especially informative. Table 3 highlights commonly occurring groups from which site-specific monitoring species may be selected, along with the types of quality changes they can help indicate.


©: Gloria Peralta

Flowers of *Limoniastrum monopetalum* (left) and *Salicornia* sp. canopy (right) at the Cádiz Bay Natural Park

Table 3. Selecting typical species for monitoring salt marshes and salt meadows. Modified from Bhuiyan et al. (2025)

Species group	Functional roles	Ecological roles	Sensitive to changes in quality
Crustaceans	Deposit feeder, herbivore, omnivore, predator, detritivore, burrower	Bioturbation, sediment mixing, nutrient cycling, erosion control, herbivory, predation, sediment stabilisation	Flooding, sea-level rise, ocean acidification, erosion
Polychaetes	Deposit feeder, predator, bioturbator	Bioturbation, nutrient cycling, sediment stabilisation	Ocean acidification, temperature shifts
Nematodes	Fungivore, microbivore	Nutrient cycling, sediment mixing	Sediment chemistry, pH
Oligochaetes	Deposit feeder	Bioturbation, nutrient cycling	Organic enrichment
Bivalves	Filter feeder, deposit feeder	Water filtration, sediment stabilisation, nutrient cycling, habitat provision	Ocean acidification, salinity, pollution
Gastropods	Grazer, detritivore, deposit feeder	Grazing, nutrient cycling, plant–microbe interactions	Ocean acidification, salinity, eutrophication
Amphipods	Detritivore, deposit feeder	Organic matter decomposition, prey for birds and fish	Ocean acidification, organic load, hypoxia
Shrimps	Omnivore, deposit/filter feeder	Trophic links, nutrient cycling, sediment turnover	Pollution, ocean acidification, habitat changes
Isopods	Herbivore, detritivore	Nutrient cycling, grazing, sediment erosion	Ocean acidification, sediment disturbance
Insects	Detritivore, herbivore, predator	Nutrient cycling, plant damage, insect population control	Habitat changes, contamination
Fishes	Omnivore, planktivore	Trophic interactions, prey for birds and larger fish	Temperature, habitat quality
Birds	Predator, omnivore, insectivore, scavenger	Trophic regulation, seed dispersal, nutrient transfer, indicators of ecosystem health	Human disturbance, habitat loss, food web changes, water quality

2. Analysis of existing methodologies for the assessment and monitoring of habitat condition

Article 17 of the Habitats Directive requires each Member State to report, among other elements, the conservation status of habitats. Conservation status must be assessed using a standard methodology and categorised as either favourable, unfavourable-inadequate, or unfavourable-bad, based on four criteria defined in Article 1 of the Directive. These criteria are (1) range, (2) area, (3) structure and functions, and (4) future prospects. The criteria for range and area are typically assessed using GIS methodologies and remote sensing techniques, while the criterion for future prospects is based on expert evaluations covering a 12-year outlook. The assessment of structure and functions criterion is based on multi-criteria analysis of monitoring data.

This section summarises the methods used by Member States to assess the structure and functions of salt marshes and salt meadows (habitat subgroups 13 and 14). This analysis is based on 19 reports from 11 MSs: Belgium (Flanders), Bulgaria, Czechia, Germany, Denmark, Spain, Ireland, Italy, Netherlands, Poland, Romania.

2.1 Variables used, metrics and measurement methods, existing data sources

The methodologies developed by the different MSs focus on similar variables, which in some cases has allowed these variables to be grouped into broader types (See Table 4). Not all variables are measured in each Member State. All 11 Member States assessed include biotic variables to estimate habitat condition, but fewer include abiotic conditions (8 MSs) or landscape/seascape properties (6 MSs).

In this chapter, we review and analyse the variables used by Member States to assess the condition of salt marsh and salt meadow habitats, and consider how these variables correspond to the ecological characterisation outlined in the previous section. We acknowledge that pressures and threats should not formally be included in the assessment of habitat condition, as these indicators are addressed under a separate parameter in the conservation status evaluation. However, because some countries use pressures and threats as if they were status indicators, their inclusion in habitat condition assessment does occur in practice. For this reason, selected examples of pressures and threats relevant to salt marshes and salt meadows are exceptionally included in this analysis (Table 4).

Table 4. Classification of variable groups according to abiotic, biotic and landscape categories relevant to salt marsh and salt meadow habitats

Category	Sub-category	Type of variable	HDSG	MSs
A1. Physical	Groundwater	Average spring groundwater level	13	BE[18], IT[2]
	Soil	Inundation frequency	13	BE[16,18], BG[11], DE[12], ES[6], RO[17]
			14	ES[9,15], RO[17]
	Anthropogenic pressure	Degree of physical alteration of the soil	13	ES[6,8]
			14	ES[9]
A2. Chemical	Air	Air nitrogen deposition	13	BE[18]
	Surface water	Salinity	13	BE[18], BG[11], PL[14]
	Groundwater	Nutrient content	13	BE[18]
		Salinity	13	BE[18]
	Soil	Nutrient content	13	BE[18], DK[1]
			14	ES[15]
		pH	13	BE[18]
		Salinity	13	BE[18], ES[6]
			14	ES[9]
	Anthropogenic pressure	Soil eutrophication and use of pesticides	13	BG[11], ES[6,8]
			14	BG[11]
B1. Compositional	Vegetation	Characteristic species	13	BE[16], CZ[13], DE[3,12], IE[5], ES[6,8], IT[2], NL[4], PL[14], RO[10]
			14	ES[9,15], IE[5], IT[2], RO[10]
		Harmful invasive species (including Invasive Alien Species)	13	BE[16], BG[11], DE[3], DK[1], ES7,[8], IE[5], IT[2], PL[14], RO[17]
			14	BG[11], IE[5], RO[17]
B2. Structural	Vegetation	Canopy height	13	DK[1], IE[5]
		Coverage (mainly area covered by characteristic species)	13	BE[16], BG[11], CZ[19], DK[1], ES[6,7,8], IE[5], IT[2], NL[4], PL[14], RO[17]
			14	BG[11], ES[9,15], IT[2], RO[17]
		Diversity (mainly species diversity)	13	ES[6,7,8]
			14	ES[9]
		Zonation (vertical and horizontal structures of vegetation distribution in function of topography)	13	BE[16,18], CZ[13], DE[3,12], IE[5], IT[2], RO[17]
			14	IE[5], RO[17]
B3. Functional	Soil	Seed bank	13	ES[6,8]
			14	ES[9]
	Vegetation	Colonisation	14	ES[15]
	Process naturalness	Functional dynamic (naturalness of physical processes)	13	BE[16]
		Ecological and geological naturalness	13	RO[17]
			14	RO[17]
	Anthropogenic pressures	Management practises (this variable is considered positive)	13	CZ[19]
		Anthropogenic impacts (mainly hydrodynamic perturbances, infrastructure and pollution)	13	BG[11], CZ[13], DE[3,12], ES[6,7,8], IE[5], RO[17]
			14	BG[11], ES[9,15], IE[5], IT[2], RO[17]
		Grazing	13	DK[1]

Category	Sub-category	Type of variable	HDSG	MSs
C1. Landscape and seascape (e.g., landscape diversity, connectivity, fragmentation)	Landscape naturalness	Landscape extent	13	BE[16], IT[2], PL[14], RO[17]
			14	IT[2], RO[17]
		Connectivity	13	BE[16], DE[12]
	Resilience	Accommodation area	13	BE[16], ES[7,8]
	Anthropogenic pressure	Fragmentation	13	BG[11]
			14	BG[1]

The "Sub-category" refers to compartment in which the variable is measured or, alternatively, to the type of anthropogenic pressure. "HDSG" indicates the Habitats Directive subgroup in which the variable is used. "MSs" shows the number of Member States that have included the variable in their methodologies. 1 (Nygaard, et al., 2020); 2 (Angelini et al., 2016); 3 (Bund-Länder-Arbeitskreis FFH-Monitoring und Berichtspflicht, 2018); 4 (van Beek et al., 2021); 5 (Brophy et al., 2019); 6 (Espinar, 2009a); 7 (Espinar, 2009b); 8 (Espinar, 2009c); 9 (Espinar, 2009d); 10 (Gafta et al., 2008); 11 (MOEW, 2013); 12 (Krause et al., 2008); 13 (Lustyk, 2023); 14 (Bosiacka, 2010, 2012; Piernik, 2012); 15 (Mota Poveda et al., 2009); 16 (Oosterlynck et al., 2020); 17 (Trif et al., 2015); 18 (Calster et al., 2020); 19 (Vydrová et al., 2014).

2.1.1 Abiotic variables for habitat condition assessment

The main abiotic conditions defining the occurrence of salt marshes and salt meadows are inundation patterns, salinity, and nutrient sources. Substantial changes in these conditions can lead to shifts in dominant characteristic species and affect community stability, potentially resulting in the establishment of an alternative stable state. This highlights the importance of characterising the abiotic conditions of the system.

Only eight Member States (Belgium, Bulgaria, Germany, Denmark, Spain, Italy, Poland and Romania) use abiotic variables to assess habitat condition in salt marshes and salt meadows. It is worth noting that the methodologies developed for Romania and Bulgaria provide only limited detail on measurement methods and do not include reference values.

Abiotic variables encompass both physical and chemical aspects. The most commonly used physical variables relate to inundation patterns, such as tidal dynamics or the level and frequency of flooding. Chemical variables primarily assess salinity and nutrient status (Table 5). In some cases, MSs also include anthropogenic pressures in the abiotic characterisation, such as soil disturbance, eutrophication, and the use of pesticides (Table 5).

Salt marshes are particularly dependent on suitable conditions related to sediment supply and inundation regimes, both of which are determined by tidal and sediment dynamics. Pioneer habitats such as 1310 and 1320 depend on sediment accretion and specific inundation windows that allow for establishment (Hu et al., 2015). In contrast, habitats located higher in the tidal range, such as 1420, require shorter inundation periods to persist. Consequently, changes in mean sea level or flooding frequency can lead to a shift from 1420 to 1310 or to 1320, or affect the distribution of these habitats.

Despite the importance of sediment dynamics for habitat establishment and persistence, none of the Member States include sediment supply as a condition variable for habitat assessment in salt marshes. However, five MSs evaluate flood frequency, two measure groundwater level, and one measures soil moisture (Table 5). Unfortunately, none of the methodologies specify how these variables are measured. Some countries, such as Poland and Romania, rely on expert visual assessments to estimate flood frequency. Belgium does not describe the method used but sets a threshold of 50-85% flooding as a favourable condition. Spain assesses flooding by recording the number of days that water stands above the ground surface.

Table 5. Summary of abiotic characteristics used for habitat condition assessment under Annex I of the Habitats Directive

Variable group	Variable	Habitat - Country(*)	Reference values with metrics where available
Physical properties	Groundwater level	1310[BE(B), IT(B)], 1320[IT(B)], 1330[BE(B)]	-0.13 – 0.41 (m – mv)[18] or WFD thresholds[2]
	Flood frequency	1310[BE(B), RO(B), ES(A), BG(B)], 1320[BE(A,B), DE(B)], 1330[BE(B)], 1410[RO(B), ES(A)], 1420[ES(A)]	1310 & 1320: 50– 85% of the tides[18], or indication for establish according to the reference ecosystems[16].
	Soil moisture	1430[ES(A)]	FV[15]: 4 – 20% W, 4 – 15% S. U1[15]: 20 – 30% W, 15 – 25% S. U2[15]: > 30 W and 25 S
Variables that evaluate salinity and nutrient conditions	Salinity	Surface water: 1310[PL(B),BE(A),BG(U)], 1320[BE(B)], 1330[PL(B)], 1340[PL(B)]	1310 & 1330 (FV: Natural high salinity, U1: Weakened, U2: Strongly weakened)[14] 1320 (FV: 20 – 30)[18] 1340 (FV: EC > 4 dS m^{-1}, U1: Altered, U2: Strongly limited)[14]
		Ground water: 1310[BE(B)], 1320[BE(B)], 1330[BE(B)]	1310 and 1320 (FV: >3000 mg Cl L^{-1})[18] 1330 (FV: 3000 – 10000 mg Cl L^{-1})[18]
		Soil: 1310[BE(B),ES(B)], 1410[ES(B)], 1420[ES(B)], 1430[ES(A)]	1310 (FV: Maintenance of natural dynamics[6] or 0 – 70 mg Cl kg^{-1})[18] 1430 (FV: EC 1 – 7 dS m^{-1}, U1: EC 7 – 10 dS m^{-1}, U2: EC > 10 dS m^{-1})[15] 1410, 1420: values must be established based on the study of reference ecosystems[9].
	N, P, SO_4, Na, K, Ca, Mg, etc	Groundwater: 1310[BE(U)]	Not provided[18]
		Soil: 1310[BE(B), DK(B)], 1330[BE(A)], 1340[DK(B)], 1430[ES(U)]	1310 & 1340 (FV: Ellenb'rg's nutritional ratio < 0.85)[1] 1330 (FV: <70 mg P kg^{-1}, 8 -43 ratio Fe:P, 8.7 – 21 C:N ratio, 2 – 9 N:P ratio; 15 – 54 sum of exchangeable cations, 7 – 38 exchangeable calcium)[18] 1430 (not specified)[15]
Other chemical properties	Soil pH, Air nitrogen deposition	Soil pH: 1330[BE(A)] Air N deposition: 1310[B(B)], 1320[B(B)], 1330 [B(B)]	Soil pH. 1330 (FV: 6.4 – 8.1)[18] Air N deposition:(FV <23 kgN ha^{-1} y^{-1})[18]

Each entry specifies the country and the level of methodological detail for the respective habitat. Reference values are included where available, with corresponding sources indicated by superscript numbers. WFD: Water Framework Directive. 1 Methodological completeness:A = Quantitative data; B = Categorical data or values derived from literature; U = Undefined. W = WINTER, S = Summer. References : 1 (Nygaard, et al., 2020); 2 (Angelini et al., 2016); 3 (Bund-Länder-Arbeitskreis FFH-Monitoring und Berichtspflicht, 2018); 4 (van Beek et al., 2021); 5 (Brophy et al., 2019); 6 (Espinar, 2009a); 7 (Espinar, 2009b); 8 (Espinar, 2009c); 9 (Espinar, 2009d); 10 (Gafta et al., 2008); 11 (MOEW, 2013); 12 (Krause et al., 2008); 13 (Lustyk, 2023); 14 (Bosiacka, 2010, 2012; Piernik, 2012); 15 (Mota Poveda et al., 2009); 16 (Oosterlynck et al., 2020); 17 (Trif et al., 2015); 18 (Calster et al., 2020); 19 (Vydrová et al., 2014).

All the above indications are valid for estimating inundation patterns; however, they appear insufficient to assess habitat resilience under sea level rise (SLR) scenarios. For a preliminary estimation of resilience, it is essential to measure elevation in relation to a known vertical datum and to account for the local tidal range. In this regard, Belgium recommends measuring flooding levels as water level height relative to mean high water (GHW), and consider values within 1 metre above or below this threshold as favourable. However, the methodology does not specify how or over what time scale these measurements should be taken.

For habitats disconnected from the tidal regime (e.g. 1430), variables such as soil humidity (%) and average spring groundwater level (m) may be more relevant than inundation frequency. Soil humidity is recorded by one MS (Spain), although no threshold values are indicated. This variable is calculated as the difference between wet and dry weight, by drying the soil sample at 50°C to constant weight.

Average spring groundwater level (m) is described by two MSs, Belgium and Spain. In habitats without tidal influence, groundwater should be near or slightly above ground level in the winter and remain hight into spring. During summer, groundwater may recede, but only slightly. To measure ground water level, Spain recommends the installation of piezometers (i.e., vertical PVC pipes perforated at the base and capped at the top). To take a measurement, the cap is removed and a measuring tape or suitable sensor is inserted to record water depth.

Belgium notes that favourable groundwater level ranges remain undefined. However, using piezometer measurements, they report values of 0.32 – -0.24 m and 0.27 – 0.11 m for the mean lowest and mean highest groundwater levels, respectively. No values are specified for the mean groundwater level.

Methodologies available from Belgium (Flanders), Bulgaria, Denmark, Spain and Poland describe the use of chemical variables for habitat condition assessment. The main chemical variables included are those related to salinity and nutrient content. Water salinity has a significant influence on species composition and the structural characteristics of different habitat types. It is a key factor controlling productivity and zonation in both coastal and non-coastal halophytic habitats (Schoolmaster & Stagg, 2018). Consequently, changes in salinity can affect the successional stage of the vegetation.

The five MSs measure salinity as sodium chloride (NaCl) concentration or as chloride content. Belgium provides threshold values based on chloride concentrations (3000 – 10000 mg L-1, Table 5). Spain proposes measuring salinity through electrical conductivity, while Poland relies on visual assessment, and Bulgaria does not specify the method. Salinity is measured in surface water, soil, and/or groundwater, with soil and groundwater salinity considered more relevant in areas with limited or no tidal influence.

The second major group of chemical variables relates to nutrient content. This group includes a range of soil properties, such as concentrations of Ca^{2+}, Na^{+}, Cl, and P, as well as exchangeable cations and nutrient ratios including Fe:P, N:P, and C:N (see Table 5). Nutrient content is considered both a condition necessary for plant development and an indicator of eutrophication, as excess nutrient availability is detrimental to the structure of halophyte communities (Johnson et al., 2016). Spain recommends monthly sampling to determine concentrations of dissolved nutrients, including nitrate, ammonium, and phosphate. Samples should be taken from both water and sediment, using standardised protocols.

No specific extraction protocols are indicated for nutrient content, except for Ellenberg's nutrient index and the Olsen-P Method. Denmark is the only Member State that applies Ellenberg's nutrient index. Ellenberg's indicator values are based on a simple ordinal classification of plant species according to the position of their realised ecological niche along environmental gradients, such as nutrient availability. By calculating the average of Ellenberg's values for all species within a sample plot, it is possible to derive an indication of the prevailing environmental conditions.

Denmark uses the nutrient ratio, defined as the ratio between Ellenberg N (nutrient) and Ellenberg R (pH) values. This ratio is used as an indicator of nutrient load in many grass- and herb-dominated habitat types, where natural variation in pH is expected. In fact, variation in

Ellenberg's indicator values for nutrients appears to be largely correlated with pH differences, as species that favour nutrient-rich environments also tend to prefer high pH conditions (Andersen et al., 2013; Ejrnæs et al., 2009). For salt meadow habitats, such as 1310 and 1340, where relatively high nutrient levels are natural, favourable values of the nutrient ratio are high (up to 0.85; see Table 5).

Belgium is the only Member State that uses the Olsen-P method, although no description of the method is provided in national reports. The method is based on the extraction of phosphate from the soil using a 0.5 N sodium bicarbonate solution adjusted to pH 8.5, applied to 5 g soil samples (Extractable Phosphorus – Olsen Method, n.d.). In addition, Belgium assesses soil pH and atmospheric nitrogen (N) deposition when evaluating habitat condition. Favourable soil pH values are only defined for habitat 1330, with a of range 6.4 to 8.1 (Table 5). Airborne nitrogen deposition is considered an additional source of eutrophication, with values below 23 kg N ha-1 y-1 deemed favourable for habitats 1310, 1320 and 1330 (Calster et al., 2020).

2.1.2 Biotic variables for habitat condition assessment

All Member States evaluating salt marshes and salt meadows include biotic variables in their habitat condition assessments (Belgium, Bulgaria, Czechia, Denmark, Estonia, Ireland, Italy, the Netherlands, Poland, Romania) (Table 6). The biotic characterisation can be grouped into compositional, structural, and functional properties. Among these, compositional and structural variables are included by more MSs (11) than functional ones (9). Most compositional and structural variables focus on the presence or coverage of characteristic species and are used to detect zonation patterns, identify changes and transitions between successional stages, or indicate habitat degradation. The establishment of zonation patterns plays a crucial role in these highly dynamic habitat types and often results in transitional forms, either between different habitat types or within a single habitat type.

The 11 Member States that have described compositional variables include Belgium (Flanders), Bulgaria, Czechia, Germany, Denmark, Spain, Ireland, Italy, Netherlands, Poland, and Romania (Table 6). Compositional variables cover the presence of positive species (e.g. key species), as well as negative species (e.g., harmful invasive species, including invasive alien species). Most MSs do not provide specific species names, but recommend compiling local species list or evaluating changes in biodiversity. Those MSs that have included species lists (e.g. BE, PL, ES, IE) consider the presence or a certain threshold abundance – of these species as indicative of a favourable condition. The main method used to characterise biological composition is counting the number of plant species present, followed by measuring the area covered by characteristic species (and HIS). In some cases, the assessment is based on expert visual evaluation.

Structural variables include canopy height, plant coverage (mainly of characteristic species), diversity, and the presence of zonation patterns. Canopy height reflects variation in vegetation structure and physiology, with the aim of maintaining structural heterogeneity within each habitat. Spain includes plant height as part of the characterisation of control points, whereas Ireland uses the standard deviation of the median maximum leaf height, measured in four 2 x 2 m plots.

Vegetation coverage is likely the most commonly used variable for assessing habitat condition, reported by 10 MSs. It is expressed as a percentage and serves as a proxy for population density, applied to both positive and negative species. Methods for evaluating coverage range from expert visual assessment (e.g., in Poland) to remote sensing analysis (as in Spain).

Table 6. Summary of biotic characteristics used for condition assessment by habitat type under Annex I of the Habitats Directive

Variable group	Variable	Habitat - Country(*)	References with units where available
Compositional	Characteristic species	1310[BE(A),ES(A)(A),IE(A),IT(A),NL(A),PL(A),RO] 1320[BE(A),DE(A),IT(A),NL] 1330[BE(A),ES(A),DE(A),IE(A),NL(A),PL] 1340[CZ(A),DE(A),IT] 1410[ES(A),IE(A),IT(A),RO] 1420[ES(A),IE(A),IT] 1430[ES(A),IT(A)]	1310 (FV: > 2 characteristic species[16]; 10% area cover[16]; >90% cover with characteristic species[16,14]; specific list of characteristic species[6]) 1320 (FV: Presence of *S. maritima*[12,16]) 1330 (FV: >30% cover with characteristic species[16]; specific list of characteristic species[7]; >10 characteristic species present[4,5]) 1340 (FV: Presence of characteristic species)[3] 1410, 1420 & 1430 (FV: Presence of characteristic species or no evidence of biodiversity decline[5,9,10,15])
	Harmful invasive species (including Invasive Alien Species)	1310[BE(B),BG(B),DK(B),IE(B),PL(B),RO(B)] 1320[ES(B),IT(U)], 1330[BE(B),ES(B),IE(B),PL(B)] 1340[BG(A),DE(A),DK(U),PL(B)] 1410[BG(U),IE(U),RO(U)],1420[IE(U)]	1310 (FV: <30% HIS[11,16]) 1330 (FV: Very variable depending on the MS; ≤ 70% reeds[16]; < 10%[14]) 1340 (FV: <5%[3]; <10%[14]; <30%[11])
Structural	Canopy height	1330[IE(A)] 1340[DK(U)]	1330 (FV: Standard deviation >5)[5] 1340 (FV: < 20 or 50[1])
	Coverage (mainly characteristic species)	1310[BE(B),BG(A),DK(B),ES(U),IE(A),IT(U),NL(U),PL(B),RO(U)] 1320[ES(B),NL(U)],1330[ES(U),NL(U),PL(B)] 1340[BG(A),CZ(B),DK(B),PL(B)] 1410[BG(U),ES(U),IT(U),RO(U)],1420[ES(U),IT(U)] 1430[ES(B),IT(U)]	1310(FV: >90%[16,11,1]; >75%[14]; >10 plants m^{-2} [(5)]) 1330 (FV: >75%[14]) 1340 (FV: >90%[11,1]; >75%[14]) 1430 (FV: >75%[15])
	Diversity	1310, 1320, 1330, 1410, 1420 (ES)	Not defined
	Zonation	1310[BE(B),RO(U)],1320[BE(B),DE(B)] 1330[BE(B),DE(B),IE(B)],1340[CZ(U),DE(U),IT(U)] 1410[IE(U),RO(U)],1420[IE(U)]	1310 &1320 (FV: mosaic, adjacent to 1330 or 1310/1320[16]) 1330 (FV: Natural zonation[12])
Functional	Seed bank	1310, 1330, 1410, 1420 (ES)	Not defined
	Colonisation	1430[ES(A)]	FV: >5 spawners plot^{-1} [(15)]
	Process naturalness	1310[RO(B),BE(B)] 1410[RO(B)]	Not defined
	Management practices	1340[CZ(B)]	Not defined
	Anthropogenic pressures	1310[ES(B),IE(B),RO(U)],1320[DE(B),ES(B)] 1330[DE(B),ES(U),IE(B)],1340[BG(B),CZ(B),DE(B)] 1410[BG(U),ES(U),IE(B),IT(B),RO(B)] 1420[ES(U),IE(B),IT(U)],1430[ES(U),IT(U)]	FV: Without land modifications

For each habitat, the Member State and the methodological completeness are indicated. Reference values are included where available. References are provided as superscript numbers next to the corresponding values. *Methodological completeness: A – Quantitative data; B – Categorical data or data derived from literature; U – Unclear or undefined. HIS: Harmful invasive species.References: 1 (Nygaard, et al., 2020); 2 (Angelini et al., 2016); 3 (Bund-Länder-Arbeitskreis FFH-Monitoring und Berichtspflicht, 2018); 4 (van Beek et al., 2021); 5 (Brophy et al., 2019); 6 (Espinar, 2009a); 7 (Espinar, 2009b); 8 (Espinar, 2009c); 9 (Espinar, 2009d); 10 (Gafta et al., 2008); 11 (MOEW, 2013); 12 (Krause et al., 2008); 13 (Lustyk, 2023); 14 (Bosiacka, 2010, 2012; Piernik, 2012); 15 (Mota Poveda et al., 2009); 16 (Oosterlynck et al., 2020); 17 (Trif et al., 2015); 18 (Calster et al., 2020); 19 (Vydrová et al., 2014).

In some cases, the coverage of woody vegetation, shrubs, wheat grasses, or reed development is used as an indicator of abiotic change (e.g. reduced flooding frequency, lower salinity) or management practices (e.g. absence of grazing or overgrazing). These changes may lead to the displacement of key halophytic species.

Although many Member States estimate the presence of characteristic species, Spain is the only MS that explicitly includes species and genetic diversity as a proxy for community structure (Espinar, 2009b). The method involves direct sampling in 0.5 × 0.5 m quadrats. However, no favourable reference values are specified.

The final group of variables used to characterise structural components in habitat condition assessments relates to zonation. Zonation refers to the pattern of spatial distribution. In coastal areas, this pattern often forms parallel bands to the MSL, primarily influenced by inundation and salinity gradients, with corresponding effects on both horizontal and vertical distributions.

Within the zonation group, variables include those that estimate patchiness as well as horizontal and vertical gradients. For example, in habitat 1310, the presence of adjacent habitats such as 1330, 1320 or 1140 is evaluated. These are all considered natural habitats forming part of the sea – land gradient of low-lying coastal areas. In such cases, horizontal and vertical structures are linked to the tidal regime, which in turn affects the corresponding abiotic conditions (i.e., inundation frequency, salinity, erosion/sedimentation processes).

In the case of coastal habitats, zonation and ecological succession are closely related (Balke, 2013). The methodology used in the Netherlands describes zonation as the presence of pioneer, low, medium, and climax stages or bands, arranged from sea to land (Janssen et al., 2020). The favourable criterion is that no single zone should cover more than 35% (or 40%) of the total area, and no less than 5%. In addition, the proportion of climax vegetation must not exceed 50% of the high or the brackish zone. The order of the bands or stages is also important, as reverse zonation – with pioneer species located in the high marsh – may indicate erosion. Spain recommends annual in situ sampling to assess zonation patterns.

Nine MSs have described functional variables, including Belgium (Flanders), Bulgaria, Czechia, Germany, Denmark, Spain, Ireland, Italy, and Romania (Table 4). Functional variables aim to evaluate the ecological processes that support the persistence of the system. These include variables assessing natural processes (e.g., seed bank, colonisation) as well as anthropogenic influences, such as management and pressures (Table 6).

Spain is the only MS that includes the assessment of the seed bank and the potential for colonisation (Table 6). The seed bank is included as a proxy for potential species diversity, measured through direct sampling and expressed as diversity indices. A more diverse and abundant seed bank indicates a better conservation status, provided that the seeds belong to species characteristic of the habitat. Thresholds must be established based on the study of reference sites and the relative importance of sexual reproduction mechanisms in the species constituting the habitat (Espinar, 2009a, 2009c, 2009d, 2009e).

Colonisation is only indicated for habitat 1430, where direct measurements of the number of breeding individuals in established permanent plots are recommended. Favourable conservation status is defined as a lambda (λ) value greater than 1 (Mota Poveda et al., 2009).

Process naturalness includes the evaluation of functional dynamics of the habitat (Belgium; Oosterlynck et al., 2020) and the naturalness of ecological and geological conditions in halophytic habitats (Romania; Trif et al., 2015) (Table 4). The assessment of functional dynamics includes a combination of natural inundation frequency (e.g., flooding during storms and drought), allowance for moderate grazing and treading, and an evaluation of geological conditions. These factors aim to assess the conditions necessary for the habitat to establish and function. No further explanation of measurement methods or reference values is provided.

Finally, eight MSs include variables to evaluate the effects of anthropogenic actions on habitat functionality (Table 6). Most of the anthropogenic impacts assessed relate to the naturalness of inundation processes, the presence of coastal infrastructure, and changes in land use. Less frequently included variables address ground disturbances, grazing intensity, anthropic discharges, and pollution.

In most cases, reference values are either qualitative, based on expert visual estimations, or undefined. Exceptionally, Czechia includes the assessment of management practices as a positive indicator; however, no reference values are specified.

2.1.3 Landscape variables for condition assessment

Landscape variables are essential for understanding the influence of regional-scale processes on habitat condition. Seven MSs have described landscape variables, including Belgium, Bulgaria, Germany, Spain, Italy, Poland, and Romania (Table 7). This characterisation includes processes relevant to the spatial distribution of habitats and their connectivity. Specifically, the variables used in Member States habitat condition assessments have been grouped into landscape naturalness (including estimation of landscape extent and connectivity), resilience, and anthropogenic pressures at the landscape level (mainly fragmentation).

Landscape extent is important because larger habitats are generally more resilient than smaller ones, particularly in salt marshes, where long-term retreat and expansion cycles have been documented (Bouma et al., 2016). Anthropogenic fragmentation poses a clear threat not only to landscape extent in salt marshes and salt meadows (Aranda et al., 2022; European Environment Agency & Schweiz, 2011), but also to connectivity, which in coastal habitats is especially important for maintaining fluxes of matter and energy and supporting colonisation potential (Wang et al., 2021).

One Member State, Belgium, has developed a methodology to assess spatial cohesion as a means of evaluating landscape extent and connectivity. The approach is based on island theory (MacArthur & Wilson, 1967), in which the size, number, and distance between habitat patches determine the likelihood of habitat persistence. The key working unit in this methodology is the functional habitat cluster – a collection of biotopes that together provide the necessary resources for a set of characteristic species of a specific habitat type. In other words, a functional habitat cluster is a group of related biotope types, situated within a bridgeable distance for characteristic species, and collectively fulfilling their habitat requirements. The methodology evaluates both the area and shape of each functional habitat cluster, as well as its distance (i.e. connectivity) to neighbouring clusters. However, basic data and reference frameworks to define threshold values for shape or distance are still lacking. As a result, current assessments are limited to a surface-based evaluation of the functional habitat clusters. Other

Member States assessing landscape extent typically rely on time series analyses of habitat mapping to estimate the areas occupied by habitats. In all cases, a favourable condition is associated with an increase in both extent and connectivity (Table 7).

Table 7. Summary of landscape/seascape characteristics used for condition assessment by habitat type under Annex I of the Habitats Directive

Variable group	Variable	Habitat - Country(*)	References with units where available
Landscape naturalness	Landscape extent (dimension of the landscape)	1310[BE(B),PL(B),RO(B)], 1320[BE(B),IT(B)], 1330[BE(B),PL(B)], 1340[PL(B)],1410[RO(B)], 1420[IT(B)]	FV: Increase in area[14] (>5 ha [16])
	Connectivity	1310[BE(B)], 1320BE[(B)], 1330[BE(B),DE(B)]	FV: >30 ha only in essential SPA areas; branched network; natural microtopography[16]
Resilience	Accommodation area	1320[BE(B),ES(B)], 1330[ES(B)]	FV: >50% has no buildings in front of the high tide[7,8]
Anthropogenic pressures	Fragmentation	1310[BG(U)], 1410[BG(U)]	FV: New fragmenting anthropogenic structures occupy up to 1% at the biogeographical level[11]

*Methodological completeness: A – Quantitative data; B – Categorical data or data derived from literature; U – Unclear or undefined. FV: favourable condition. Reference values are included where available. In cases where multiple reference values exist, the corresponding source is indicated by a numeric code. References: 1 (Nygaard, et al., 2020); 2 (Angelini et al., 2016); 3 (Bund-Länder-Arbeitskreis FFH-Monitoring und Berichtspflicht, 2018); 4 (van Beek et al., 2021); 5 (Brophy et al., 2019); 6 (Espinar, 2009a); 7 (Espinar, 2009b); 8 (Espinar, 2009c); 9 (Espinar, 2009d); 10 (Gafta et al., 2008); 11 (MOEW, 2013); 12 (Krause et al., 2008); 13 (Lustyk, 2023); 14 (Bosiacka, 2010, 2012; Piernik, 2012); 15 (Mota Poveda et al., 2009); 16 (Oosterlynck et al., 2020); 17 (Trif et al., 2015); 18 (Calster et al., 2020); 19 (Vydrová et al., 2014).

One Member State (Bulgaria) specifically measures habitat fragmentation as the proportion of area occupied by anthropogenic structures. Other MSs also quantify the presence of such structures, but in the context of assessing biotic conditions (Table 6).

Finally, only two Member States (Belgium and Spain), assess the presence of accommodation areas – adjacent areas without anthropogenic structures that have the potential to be colonised. This variable serves as a proxy for habitat resilience and, in the case of Spain, is to be evaluated using remote sensing technologies.

2.2 Definition of ranges and thresholds to obtain condition indicators

In general, the criteria for establishing thresholds that determine habitat condition (i.e. structure and functions) are insufficiently documented. In most cases, these thresholds are based on expert judgment and evaluated qualitatively (e.g., tidal creek networks: favourable = natural and meandering; unfavourable = uniform relief).

Most Member States use a multicriteria approach to assess the status of structure and functions at the local scale. However, there is considerable variation in how these criteria are ranked, particularly in the number of variables considered (see section 2.1), the thresholds applied, and the method of aggregation.

Member States can be broadly categorised according to the number of thresholds used to assess habitat structure and functions:

- **One threshold** (e.g. Ireland): A single target (threshold) is defined for each criterion. The overall assessment of structure and functions is then based on the number of failed criteria. Favourable: 0 criteria failed; Unfavourable-inadequate: 1-2 criteria failed; Unfavourable-bad: 3 or more criteria failed.
- **Two thresholds** (e.g. Denmark): Both strict and relaxed thresholds are used. Each criterion is compared against these two thresholds. If the value is better (higher or lower depending on the criterion) than the strict threshold, then indicator is considered favourable. If the value is worse than the relaxed threshold, it is unfavourable-bad. Values falling between the two thresholds are considered unfavourable-inadequate.
- **Three ranges** (e.g. Germany, Spain): Criteria are assessed using a three-level decision matrix. The evaluation is qualitative and requires expert judgement. In the case of Germany, structure and functions are assessed using three groups of variables: habitat structure, characteristic species, and disturbances. Each variable is assigned a score: A (excellent), B (good), or C (poor). The final assessment is based on the most frequently assigned score across the variables. However, if any variable is rated C, the overall assessment cannot be classified as A. Aggregation is first carried out within each group, then across the three groups. In the case of Spain, each criterion is assigned an integer value: 2 (favourable), 1 (unfavourable-inadequate), or 0 (unfavourable-bad). The overall condition of structure and functions is calculated as the average of the individual criterion scores.

Belgium evaluates structure and functions based on four groups of variables: vegetation, habitat structure, disturbances, and spatial cohesion. The number of criteria used to assess each group depends on the specific habitat type, with thresholds derived from existing literature and supplemented by expert knowledge where necessary. These thresholds for determining favourable or unfavourable conditions are calculated by applying the BioHAB/EBONE methodology to each habitat type and its ecological characteristics (Bunce et al., 2005, 2008, 2011).

The Netherlands evaluates structure and functions using a method developed under the Water Framework Directive (WFD), which assesses the extent to which a balanced zonation of salt marsh horizons is present. This includes zones such as pioneer, low, medium, climax high, and climax brackish. Based on the number of criteria met, the WFD assigns one of several scores: good (equivalent to a reference value, P-REF), moderate, inadequate or poor. The first two scores (good and P-REF) are translated to favourable (green) under the Habitats Directive; moderate is interpreted unfavourable-inadequate (orange); and inadequate and poor are classified as unfavourable-bad (red). The final score for structure and functions is then weighted by surface area and scaled up to produce a national-level assessment (Janssen et al., 2020).

2.3 Aggregation methods at the local scale

The assessment of the structure and functions criterion is based on multi-criteria evaluations of monitoring data. This section summarises the aggregation methods used by Member States to assess the structure and functions of salt marshes and salt meadows at the local scale. Methods used for aggregation at the biogeographical scale are presented in the following section.

The overall assessment at the local scale requires the integration of abiotic, biotic, and landscape variables measured in the field. This integration can be carried out using different approaches: the ‘one-out, all-out’ rule, averaging or additive methods, or a combination of these at different hierarchical aggregation levels (Langhans et al., 2014).

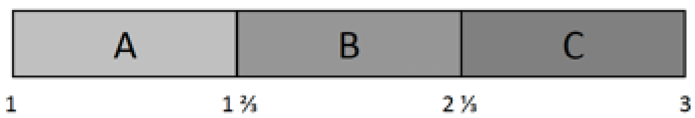
Although all 11 MSs have developed detailed methodologies to define and measure the variables used to characterise the structure and functions criterion, not all of them specify a clear aggregation method for local-scale assessments.

Germany evaluates the structure and functions criterion based on three groups of variables: (1) habitat structures, (2) species inventory, and (3) impairments (Kroiher et al., 2017). Each group is assessed qualitatively using categorical values A, B, or C, corresponding Excellent, Good or Bad condition, respectively (Krause et al., 2008).

Two strategies are used to aggregate these values into a single assessment:

- **Calculations according to the Pinneberg scheme:** This method aggregates the ordinal categories from the three variable groups using the most frequent category (e.g. (A, A, B results in A). However, in cases A, A, C and A, B, C, the result is downgraded to category B (Kroiher et al., 2017).
- **Numeric calculation**: In this method, the ordinal categories A, B, and C are assigned numerical values 1, 2, and 3, respectively. The average of these values is calculated, and the resulting mean is then converted back into an ordinal category based on predefined thresholds (as shown in Figure 3).

Figure 3. Method used for ordinal aggregation of the structure and functions criterion at the local scale in Germany



Numeric values are averaged and converted to categories A, B, or C based on the threshold ranges shown
Source: Kroiher et al., (2017).

The methodology used in Spain shares similarities with that described for Germany. The local assessment of habitat condition for salt marshes and salt meadows begins with the definition of general attributes characterising the structure and functions of halophytic habitats (Table 8). Most of these attributes are suitable for a general characterisation of salt marsh and salt meadow habitats, and their inclusion in the assessment is mandatory. In addition, habitat-specific variables may be included. For example, a specific variable related to benthic macrofauna is incorporated into the assessment of habitat 1140 (Mudflats and sandflats not covered by seawater at low tide).

Each general and specific variable is evaluated using a three-level scale: 'favourable', 'unfavourable-inadequate', or 'unfavourable-bad' (Table 8). These variables are then aggregated using the Total Vulnerability Index (TVI, Equation 2), which is calculated as the average of the Partial Vulnerability Indexes (PVI; Equation 1). PVI is computed separately for general and specific variables. It is derived from the numeric values assigned to the ordinal categories: 2 for favourable, 1 for unfavourable-inadequate, and 0 for unfavourable-bad (Table 8). The PVI is calculated as the sum of the obtained scores divided by the sum of the maximum possible scores (Equation 1).

Equation 1

$$PVI_x = \frac{\sum_1^n v_i}{\sum_1^n 2}$$

Equation 2

$$TVI = {(PVI_{General} + PVI_{Specific})}/{2}$$

Where v_i is the numeric value (2, 1, or 0) assigned to each variable *i*, and *n* is the total number of variables included for each type of variables (*x*). The resulting Total Vulnerability Index (TVI) ranges from 0 to 1, with values closer to zero indicating lower system resilience (Table 9).

Table 8. Spanish example of numeric values assigned to variables during the assessment of habitat condition in salt marshes and salt meadows

Type	Category	Variable	Favourable	Unfavourable - inadequate	Unfavourable-bad
General variables	Abiotic	Physic-chemical characteristics	2	1	0
		Abiotic perturbation	2	1	0
	Biotic	Characteristic species distribution	2	1	0
		HIS	2	1	0
		Canopy height	2	1	0
		Zonation	2	1	0
	Landscape	Natural drainage system	2	1	0
	Other	Peculiar area indicators	2	1	0
Specific variables	Habitat 1140	Benthic macrofauna	2	1	0

Source: Adapted from Aranda et al., (2019)

Table 9. Characterisation of local habitat condition for salt marshes based on the range of values of the Total Vulnerability Index (TVI) used in Spain

	Favourable	Unfavourable - inadequate	Unfavourable-bad
TVI	1.00 - 0.67	0.66 - 0.33	0.32 - 0

Source: Aranda et al., (2019)

Belgium (Flanders) assesses the structure and functions of habitats using variables grouped into four categories: (1) vegetation, (2) habitat structure, (3) disturbance, and (4) spatial cohesion. Local assessments of favourable or unfavourable condition require the aggregation of these variables.

The traditional aggregation method is the 'one-out, all-out' rule, which can be applied at the level of category criteria, allowing for some nuance in the overall evaluation. An alternative approach is based on a majority rule, in which the condition is determined by whether most indicators are favourable or unfavourable. In this case, individual indicators may be weighted according to their importance. The most heavily weighted indicators are those that (1) pose a threat to the long-term favourable condition of the habitat, and (2) are (nearly) entirely influenced by management.

Ireland assesses structure and functions at the local scale for habitats 1310, 1330, 1410, and 1420. Each habitat is evaluated using a different number of variables, with each variable having a defined reference criterion for favourable condition. The local-scale assessment is based on the number of variables that fail to meet these criteria: the condition is classified as favourable when all variables meet their respective criteria, unfavourable–inadequate when one or two variables fail, and unfavourable-bad when three or more variables fail.

Denmark applies two thresholds to determine whether a variable is favourable, unfavourable-inadequate, or unfavourable-bad. However, the method for aggregating these indicators at the local scale is not clearly defined.

The Netherlands applies the 'one-out, all-out' strategy for aggregating variables., but it remains unclear how this approach is implemented at the local scale.

2.4 Aggregation at biogeographical scale

Most Member States follow the recommendations of the Article 17 reporting guidelines for the 2013-2018 period. These guidelines define the status of the structure and functions parameter as favourable if 90% or more of the habitat area is assessed as being in good condition; unfavourable-bad if more than 25% is assessed as not in good condition; and unfavourable-inadequate for intermediate values. Accordingly, the assessment of structure and functions at the biogeographical scale is, in most cases, derived by weighting local assessments relative to the corresponding occupied area.

The overall conservation status assessment at the biogeographical level combines the evaluation of habitat condition with that of habitat range, area and future prospects. Habitat condition is usually derived by scaling up local assessments based on area. Ireland is an exception, as it uses the number of criteria failed to determine the proportion of favourable area: 100% of the site area is favourable if no criteria fail; 75% if one or two criteria fail; 25% if three criteria fail; and 0% if four or more criteria fail (Brophy et al., 2019).

In some MSs, the biogeographical-level assessment requires coordination between national and regional authorities. In Germany, for instance, dedicated conferences are held in which expert votes carry significant weight and can modify the final assessment of the habitat condition parameter in the national reports.

2.5 Selection of localities

Although most Member States have published methodological procedures for monitoring of Annex I habitats, the criteria for selecting sites where monitoring is applied are not always clearly defined or consistently established. Ireland offers a string example of a well-structured monitoring programme (Figure 4). The Irish Salt marsh Monitoring Programme (SMP) includes 61 sampling points distributed evenly along the coast, supplemented in the most recent reporting period by 24 additional sites (Brophy et al., 2019).

Field monitoring is conducted within 2 x 2 m plots, although certain criteria are assessed at broader spatial scales, including the polygon scale (i.e., Irish vegetation community scale) and the site scale. The site scale refers to inventory sites or areas identified as potential salt marshes through the interpretation of aerial photography, with site sizes ranging from 2 ha to 325 ha (Brophy et al., 2019).

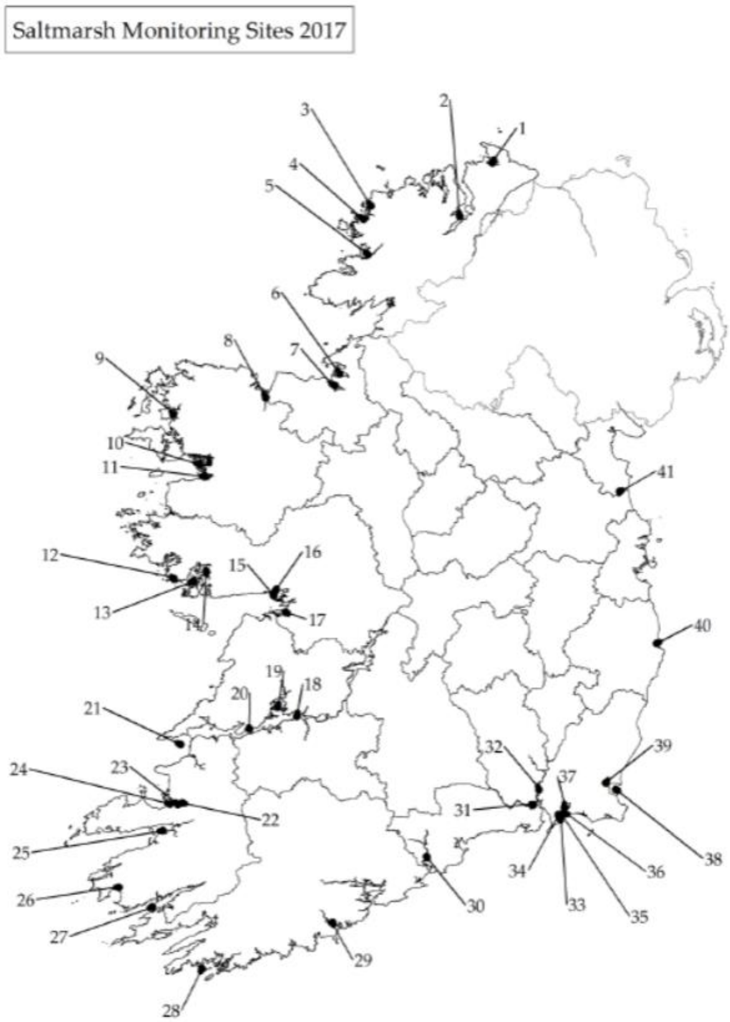
Belgium has clearly defined procedures for selecting sampling locations. Using the national habitat map as a sampling frame, 80-170 plots per habitat type are selected through a spatially balanced random distribution. The number of plots is specifically designed to minimise Type I and Type II errors. For salt marshes and salt meadows, the size of sampling plots ranges between 0.5 and 1 ha.

Germany and Italy primarily use cartographic data obtained through photointerpretation to evaluate the structure and functions of their habitats. However, this approach requires control points, the location of which is not always clearly documented. In Italy, these control points appear to follow a transect design, with the number of transects related to the extent of the habitat.

Some MSs, such as the Netherlands, use the control points from the Water Framework Directive (WFD) to assess habitat structure and functions. The Czech Republic uses 25 m^2 permanent monitoring plots, with a maximum of 50 plots per habitat. However, no specific information is provided regarding the number of plots used for salt meadow habitat type 1340.

The rest of MSs have not yet reported a monitoring programme for the regular assessment of salt marshes and salt meadows.

Figure 4. Location of sites included in the Salt Marsh Monitoring Project 2017-18 (Ireland)



Source: Brophy et al., (2019)

2.6 General monitoring and sampling methods

Most MSs have developed or adapted specific monitoring programmes to assess the status of Annex I habitats within their territory. Denmark has the National Monitoring and Assessment Programme for the Aquatic and Terrestrial Environment (NOVANA). Ireland uses the Saltmarsh Monitoring Programme (SMP), along with tools developed under the projects SMAATIE (Saltmarsh Angiosperm Assessment Tool for Ireland) and SAMFHIRES (Saltmarsh Function and Human Impacts in Relation to Ecological Status), both funded by the Environmental Protection Agency[1]. Belgium monitors salt marshes and salt meadows through the Schelde monitoring Program and the PINK Project (Permanent Inventory of Nature Reserves along the Coast, 2007-2010, 2012-2014, 2014-2018). The Netherlands has modified its water

[1] Environmental Protection Agency: https://www.epa.ie/

Framework Directive (WFD) monitoring programme to include salt marsh and salt meadow habitats. For other MSs, the monitoring programmes are less clearly defined.

In most cases, monitoring is divided into two components: (1) Sampling-based field monitoring, and (2) Cartographic assessment of designated habitat areas. Both components are typically carried out and reported at least every six years. Most MSs use standardised templates at the national level to harmonise data collection within each habitat type. Field-based monitoring is usually conducted through visual inspections at sampling points, requiring expert knowledge. Cartographic assessments are generally produced using GIS based on photointerpretation of orthophotography.

Sampling plots vary widely in both size and number between MSs – for example, Ireland uses 2 x 2 m plots (85 plots in total), while Belgium uses plots 0.5 to 1 ha (80-170 plots per habitat type). However, for many MSs, the sampling design – particularly the size and number of plots – is poorly documented.

2.7 Other relevant methodologies

Understanding the dynamics of salt marshes and salt meadows requires the application of effective methodologies to support conservation and restoration efforts worldwide. International examples include the Saltmarsh Habitat and Avian Research Program (SHARP, USA), the Atlantic Climate Adaptation Solutions Association Programme (ACAS, Canada), and the Estuarine Habitat Monitoring and Threat Assessment Project (New South Wales, Australia).

Although each country tailors its approach to local environmental conditions and policy frameworks, core methodologies tend to be consistent across regions. The most representative monitoring methodologies applied in these countries include:

- **Field-based vegetation surveys** (e.g., USA, Canada, Australia): Transects and quadrats are used to assess species composition, vegetation cover, and plant health.
- **Wildlife and biodiversity monitoring** (e.g., USA): Surveys focus on bird species and other fauna dependent on salt marshes, providing insights into habitat quality and overall biodiversity.
- **Hydrological monitoring** (all): Piezometers, tidal gauges, and water loggers are used to track changes in inundation frequency, water salinity, and hydrodynamic conditions.
- **Sediment accretion and erosion monitoring** (e.g., USA, Canada, Australia): Surface Elevation Tables (SETs) and sediment cores are employed to assess long-term changes in sediment elevation, erosion rates, and carbon storage potential.
- **Remote sensing and GIS** (all): Satellite imagery, drone surveys, and LiDAR technologies are used to map habitat extent, monitor vegetation changes, and detect erosion and accretion processes.

One of the most common challenges in monitoring salt marshes and salt meadows is the presence of soft sediments and the associated difficulties in site. In many cases, field-based surveys cause considerable disturbance to soil conditions. As a result, the number of monitoring sites in most programmes reflects a trade-off between data quality, available budget, and the extent of soil disturbance. In this context, remote sensing techniques offer complementary methodologies that help alleviate these trade-offs by reducing the need for extensive on-site data collection.

For coastal habitats, a key variable determining inundation and salinity patterns – as well as the distribution of characteristic plant species (i.e., zonation) – is the vertical position of the soil relative to the tidal regime. Low and medium marshes are typically colonised by a limited number of halophytic species (e.g., habitat types 1310 and 1320) and are particularly vulnerable to climate change impacts such as sea level rise. Unfortunately, these zones – although critically important for coastal managers, scientists, and engineers – are often inaccessible: too shallow and hazardous for most traditional bathymetric survey vessels, yet too deep to be mapped using land-based survey methods (Carvalho et al., 2017).

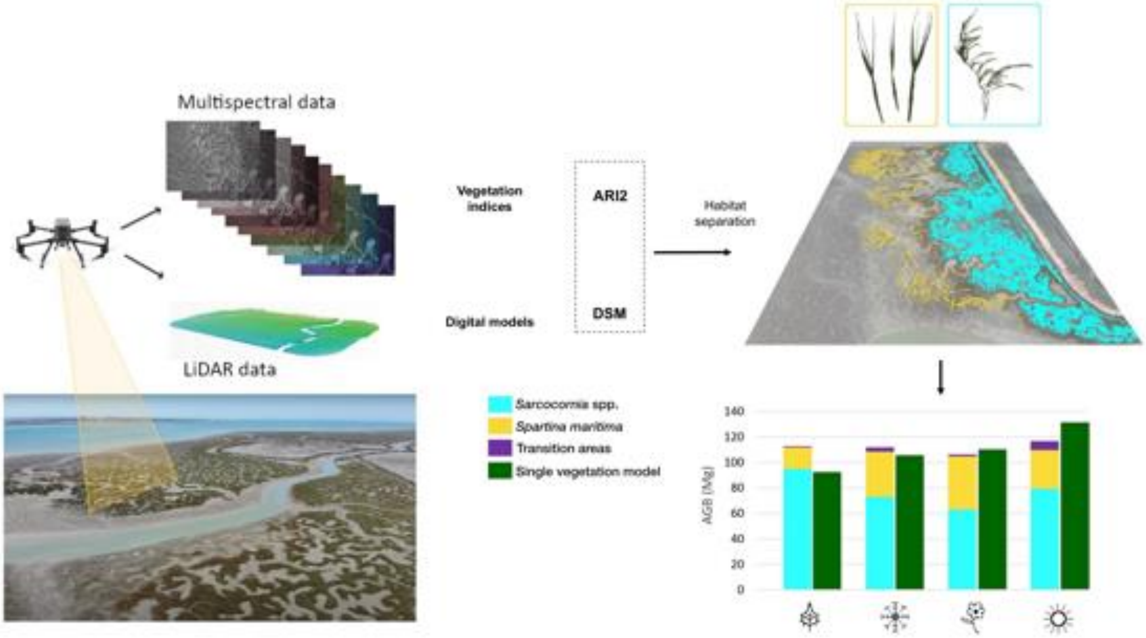
As a result, any methodological improvements in the characterisation of elevation and vegetation within the tidal zone would significantly benefit the monitoring and assessment of coastal habitat condition. In recent years, major advances have been made in the application of remote sensing techniques to characterisation of salt marshes.

UAV-based hyperspectral (HS) sensors, when combined with UAV-LiDAR, have shown the capability to monitor the distribution of halophyte species at the species level (Curcio et al., 2023). However, UAV-HS products currently require substantial resources in terms of storage space, processing time, and technical expertise. Therefore, their use is presently recommended only for specific, small-scale applications (e.g., monitoring early stages of HIS invasion).

More promising is a recent study by Curcio et al. (2024), which demonstrated the effectiveness of UAV-based sensors for monitoring habitat types 1320 and 1420 along temperate coasts. This study combined UAV-based multispectral (MS) imaging and LiDAR to monitor a 20ha salt marsh over different seasons. The results revealed distinct seasonal biomass distribution patterns for *Spartina maritima* and *Sarcocornia* spp. (Figure 5). The integration of MS and LiDAR provided Digital Terrain Models (DTMs), Canopy Height Models (CHMs) and biomass estimations – all of which are considered valuable parameters in the assessment of tidal salt marsh habitat condition.

Multispectral satellite imagery has also proven useful for characterising intertidal digital elevation maps when combined with hydrodynamic modelling (González et al., 2023), and for assessing the status and trends of tidal salt marsh vegetation (Hang et al., 2024). A major limitation of satellite-based applications is the difficulty in distinguishing species distributions, primarily due to limited spatial resolution. However, this field has advanced considerably in recent years. The combination of UAV and satellite data appears to be a promising approach for improving monitoring tools in coastal vegetated habitats. These techniques represent a significant step forward in monitoring coastal habitats at large spatial scales while minimising disturbance to the habitat.

Figure 5. Example of UAV-based sensor application for salt marsh monitoring



Source: Andrea C. Curcio

2.8 Conclusions

Salt marshes and salt meadows are complex, dynamic, and interconnected ecosystems shaped by unique physicochemical conditions – particularly salinity and inundation – which directly influence their vegetation composition. These distinctive characteristics explain why most Member States reporting on habitat groups 13 and 14 focus on variables related to inundation and salinity patterns (abiotic variables), the presence, distribution, and coverage of characteristic species (biotic variables), and spatial scale and connectivity (landscape variables) (see Table 7).

Despite the common focus, the assessment process faces several challenges: selecting appropriate proxies (variables), standardising methods and metrics, ensuring spatial representativeness (e.g., through control point design), and simplifying procedures for implementation. Each MS addresses these challenges in its own way, leading to diverse methodologies and metrics, which complicates cross-country comparisons.

The definition of ranges and thresholds for condition indicators also varies among MSs, Often, thresholds are poorly documented and based on expert judgment. For instance, Ireland applies a single threshold per criterion and evaluates overall condition based on the number of failed criteria. Denmark uses both strict and relaxed thresholds to categorise conditions. Germany and Spain define three condition classes; Germany applies a qualitative decision matrix, while Spain averages integer scores across criteria. Belgium bases thresholds on literature and expert opinion, and the Netherlands follows a methodology aligned with the WFD for habitat assessment.

For aggregation at the local scale, Germany employs both categorical aggregation and numerical averaging, while Spain uses the Total Vulnerability Index (TVI) to integrate structure and functions variables. Belgium combines 'one-out, all-out' approach with majority rule, placing particular emphasis on variables that reflect long-term threats to habitat condition. Ireland aggregates by counting variables that fail to meet established criteria, whereas Denmark applies threshold-based evaluation but lacks a clearly defined method for local-level aggregation. The Netherlands also uses the 'one-out, all-out' approach, though detailed procedures for local aggregation are not reported.

At the biogeographical scale, most MSs aggregate local assessments by weighting them according to the occupied area. Ireland is an exception, determining the proportion of favourable area based on the number of failed criteria. In Germany, biogeographical assessments require coordination between national and regional authorities, with expert votes playing a significant role in the final outcome.

A major advance in reporting the condition of Annex I habitats has been the development of specific monitoring programmes. However, only a few MSs have developed or adapted such programmes to assess habitat conditions within their distribution areas. For salt marsh and salt meadow habitats in particular, Denmark has established the National Monitoring and Assessment Programme for the Aquatic and Terrestrial Environment (NOVANA), while Ireland uses the Salt marsh Monitoring Programme (SMP), alongside tools developed through the SMAATIE (Saltmarsh Angiosperm Assessment Tool for Ireland) and SAMFHIRES (Saltmarsh Function and Human Impacts in Relation to Ecological Status) projects from the Environmental Protection Agency. Belgium monitors these habitats through the Schelde Monitoring Programme and the PINK Project (Permanent Inventory of Nature Reserves along the Coast). The Netherlands has adapted its Water Framework Directive (WFD) monitoring programme to include salt marshes and salt meadows. In contrast, other MSs have less well-defined monitoring systems.

The conclusion that emerges from the review of existing methodologies for the monitoring and assessment of habitat condition is the critical need for harmonised approaches to enable comprehensive and comparable assessments across Europe. Differences in the interpretation and implementation of monitoring under Article 11 of the EU Habitats Directive have led to inconsistencies in the selection of representative proxies (variables), measurement methods, metrics, and spatial representativeness (e.g., control point design) within national monitoring programmes, thereby complicating inter-MS comparisons (Ellwanger et al., 2018).

It is evident that standardised guidelines would support the integration of data and monitoring schemes at the EU scale. In this context, remote sensing techniques could play a key role. These technologies are advancing rapidly and offer numerous innovations and tools for monitoring salt marshes and salt meadows (e.g., Curcio et al., 2023, 2024; Regos & Domínguez, 2018). However, in many cases, those responsible for managing habitat monitoring schemes currently lack access to the necessary data, computing infrastructure, standardised analytical tools, and technical expertise (Ellwanger et al., 2018). Nevertheless, the inclusion of remote sensing in monitoring procedures offers clear advantages at the European level by facilitating the upscaling of habitat condition assessments and supporting the standardisation of methods and metrics.

3. Guidance for the harmonisation of methodologies for assessment and monitoring of habitat condition

3.1 Selection of condition variables, metrics and measurement methods

At present, different variables, metrics, and methodologies are used by the EU Member States to assess the condition of saltmarsh and salt meadows habitats. This variation presents challenges in terms of aligning procedures, ensuring comparability, and enabling aggregation at the European scale.

To address these challenges, this section provides basic guidelines to support the harmonisation of MSs approaches to monitoring the condition of these habitats. The aim is to promote consistency in the selection and application of variables and methods, improve the comparability of results, and support more robust and policy-relevant assessments at both national and EU levels.

To harmonise the selection of condition variables for salt marsh and salt meadow habitats, **several general principles** should be adopted:

- Any given habitat should be assessed by a consistent set of variables, regardless of its location across different MSs.
- Contextual factors specific to each MS may influence the values of variables that define good habitat condition (i.e., reference values or thresholds).
- For a given habitat type, the overall assessment of habitat condition – based on the selected variables – should be equivalent across MSs, once national contextual factors are taken into account.

Harmonising the selection of variables also requires a clear set of **common requirements**:

The description of condition variables, metrics, and measurement procedures used by each MS must be clearly communicated and fully understandable, enabling application across all Member States.

- For each salt marsh or salt meadow habitat, key types of ecological characteristics must be measured in all MSs. A core set of common variables should be monitored consistently using equivalent procedures and standardised metrics. In addition, certain habitat types may require specific variables, for example, tidal and coastal salt marshes should be assessed using tidal range to describe inundation patterns, while inland salt meadows are better characterised using spring groundwater level. A further set of recommended condition variables are recommended but may be considered optional and may not be applied in all cases, depending on resources and contextual factors specific to each Member State.
- Essential and specific variables should reflect the fundamental characteristics of salt marshes and salt meadows and serve as indicators of the habitat condition and quality.
- The number of essential common variables should be kept to the minimum required to determine habitat condition effectively.
- These variables should meet key quality criteria: validity, reliability, availability, simplicity, and compatibility (Czúcz et al., 2021b).
- Common training sessions on the measurement of condition variables should be organised for experts across Member States to support full harmonisation.

Table 10 presents a proposed list of essential, specific, and recommended variables for assessing habitat condition in salt marsh and salt meadow habitats. **Essential variables** reflect fundamental characteristics that define the core ecological nature of these habitats. They include parameters such as soil salinity, inundation patterns, the presence of characteristic species indicative of habitat distinctiveness, and indicators of habitat quality, such as the extent of harmful invasive species and the clarity of zonation patterns. **Specific variables** are intended to capture fundamental characteristics of habitat condition that are essential for some particular habitat types within the group of saltmarsh and salt meadows or in some particular ecological contexts. For example, inundation patterns are a key determinant of habitat quality in both tidal salt marshes and inland salt meadows; however, while tidal range is the appropriate variable to assess inundation in coastal systems, spring groundwater level serves this purpose in non-coastal salt meadows. In both cases, the underlying ecological property is the same, but the appropriate variable is habitat-specific. **Recommended variables** are generally relevant for habitat assessment but may not need to be measured in all contexts, depending on the particular characteristics of the habitat or region.

The proposed metrics are designed to be simple, robust, and reliably measurable – primarily at the plot level. The list is informed by the main ecological characteristics described in Section 1.2, information provided by Member States on habitat condition assessment (see section 2), and relevant scientific literature (e.g., Bouma et al., 2016; Curcio et al., 2024; Fagherazzi et al., 2020).

The main abiotic variables relevant to habitat condition assessment include elevation, inundation patterns, salinity, sedimentation/erosion, and nutrient and organic matter content – all of which strongly influence halophyte distribution. In tidal and coastal habitats, many of these variables are collinear, with elevation relative to the tidal regime playing a decisive role in shaping inundation, salinity, and sedimentation/erosion patterns (Mossman et al., 2020).

The generation of accurate Digital Terrain Models (DTMs) to characterize elevation is highly recommended, particularly in intertidal and coastal systems. Elevation relative to mean sea level, combined with the local tidal range, enables a reliable estimation of inundation patterns, which are key to evaluating habitat condition. This assessment must consider both the status of the biota and the abiotic conditions that support its presence.

In intertidal systems, such as coastal salt marshes, habitat distribution is extremely sensitive to small elevation changes, with notable shifts occurring in response to differences of just 10–20 cm, likely due to changes in inundation frequency (Curcio et al., 2024). However, the availability of high-precision DTMs in these areas remains very limited. Long-term changes in inundation patterns driven by sea-level rise occur gradually, while successional responses in the biota may take even longer (Fagherazzi et al., 2020). In this context, accurately determining the frequency and duration of inundation is essential to assess whether a habitat remains in a stable state or is at risk of transitioning to an alternative one. For this reason, a precise measurement of the position and elevation of each sampling plot is critical. Additionally, processes such as sedimentation and erosion, which directly affect elevation, may also alter inundation patterns, particularly when interacting with tidal dynamics.

In macrotidal and mesotidal coastal systems (e.g., habitat 1320), inundation patterns should be characterised relative to an explicitly stated standardised vertical datum, such as the Lowest Astronomical Tide (LAT), and the local tidal regime (Anderson et al., 2022). In contrast, in systems with limited or no tidal influence (e.g., microtidal or non-coastal habitats), groundwater dynamics become more relevant, as is the case for habitat 1340. Thus, while inundation patterns are an essential condition across all habitats in groups 13 and 14, the most appropriate variables to estimate them must be selected based on habitat type and are therefore habitat-specific.

Soil salinity is another essential abiotic variable for these habitat groups, but its cross-national and EU-wide comparability requires a standardised methodology. Additional important variables for the abiotic characterisation of habitat groups 13 and 14 include soil nutrient and organic matter content. However, in cases where there are no significant changes in vegetation composition or structural characteristics, these variables may be considered recommended rather than essential. Other recommended abiotic variables that may enrich the assessment include rainfall, storm surge incidence, and significant wave height.

The proposed essential variables for assessing biotic compositional characteristics include:

1. Characteristic species richness
2. Non characteristic plant species richness.
3. Presence of harmful invasive species (HIS), including invasive alien species (IAS)

A regional list of characteristic species for salt marsh and salt meadow habitats, as well as potential HIS and IAS, should be developed to support consistent assessment. Although harmful and exotic plant invasions are generally rare in saline environments due to their stressful physicochemical conditions, notable exceptions have been documented – for example, *Spartina densiflora* and its hybridisation with the native *S. maritima* in southern Europe (Castillo et al., 2008).

Environmental changes, such as sea-level rise or other global change drivers, can lead to habitat transformation, resulting in shifts in species composition and richness. The appearance of new species (non-characteristic) or the displacement of established ones may serve as early indicators of ecological regime shifts (Raposa et al., 2017). Conversely, a reduction in salinity – for instance, through freshwater discharges from urban stormwater – may facilitate the invasion of terrestrial plant species (Geedicke et al., 2021).

The proposed essential variables for describing biotic structural properties include:

1. Canopy height
2. Species coverage
3. Zonation patterns

Canopy height serves as an indicator of salt marsh structure and overall ecosystem health. Variations in canopy height can reflect changes in both biotic and abiotic conditions and may have implications for ecosystem functions and services (Rupprecht et al., 2015).

Species coverage provides insight into the relative importance of each species within the plant community and can reveal distribution patterns – high percentage coverage typically indicates a homogeneous distribution, while low percentage coverage suggests a patchy structure.

Zonation patterns offer information about the horizontal (spatial extent) and vertical (elevation) organisation of the ecosystem, which is particularly relevant in tidal and coastal habitats. The presence of distinct zonation bands (e.g., low, medium, and high marsh) is a positive indicator of structural integrity in salt marshes.

In addition, the presence of erosion cliffs in intertidal habitats is considered a specific variable, signalling active erosion processes. Once initiated, erosion along the marsh front tends to persist until the foreshore widens sufficiently to dissipate wave energy, thereby enabling the re-establishment of pioneer salt marsh vegetation (Bouma et al., 2016).

The proposed essential variable for describing biotic functional characteristics is:

Naturalness of connectivity

In wetlands, water movement is the primary mechanism for exchanging biotic and abiotic components within and between habitats (Ferronato et al., 2018; Zhao et al., 2021). This variable can be assessed qualitatively using three categories:

- **Favourable (a):** No anthropogenic structures or water management interventions that intercept or alter natural water movement.
- **Unfavourable-Inadequate (b):** Minor anthropogenic structures are present but do not significantly affect natural water movement.
- **Unfavourable-Bad (c):** Anthropogenic structures or management actions are present and significantly intercept and/or alter natural water movement.

Where management actions have successfully restored natural hydrological dynamics, the status may be reassessed as favourable, provided that sufficient connectivity is demonstrated.

The proposed essential variables for describing landscape characteristics include:

- Habitat extent
- Patchiness
- Slope
- Accommodation area

Patchiness is typically associated with habitat fragmentation and may indicate habitat degradation (Aranda et al., 2022). However, in pioneer zones of tidal salt marshes (e.g. habitat 1320), a patchy distribution is characteristic and should not be interpreted as deterioration unless a decline in vegetation extent is observed over time.

Slope is recommended as a proxy for spatial physicochemical gradients across the habitat.

Accommodation area refers to the adjacent space – free from anthropogenic structures – that allows for landward migration of salt marshes in response to sea level rise (Molino et al., 2021).

Table 10. Proposed essential, specific and recommended condition variables for salt marsh and salt meadow habitats, including corresponding metrics and measurement procedures

The metrics may offer multiple options, to be agreed upon by the Member States. Seasonal sampling may be reduced to annual sampling if conducted consistently at the same time each year, preferably during peak biomass). App.: Application (E: Essential, S: Specific, R: Recommended). f: Sampling frequency (6Y: every 6 years, A: annually, S: seasonally).

[1]See Table 1 for the correspondence of HD habitats

Group	Variable	Metrics	Measurement procedure	f	Intertidal[1]	Coastal[1]	Non-coastal[1]	App	Observations*
1. Abiotic characteristics									
1.1 Physical characteristics									
Inundation patterns	**Elevation relative to LAT**	m with respect LAT	Remote sensing or dGPS transects	6Y	*	*		E	An accurate DTM raster should be generated, or at minimum, elevation profiles perpendicular to the coastline.
			Plot surveys	S	*	*	*	E	Elevation must be measured within each plot.
	Tidal regime and range	Tidal data in m with respect LAT	Modelling	6Y	*	*		S	Relevant for intertidal and coastal habitats.
	Spring groundwater level	kPa	Piezometers	A			*	S	Highly relevant in non-coastal habitats.
	Rainfall	mm y^{-1}	Meteorological agencies	A	*	*	*	R	These environmental variables are relevant to interpret changes in the habitat condition, for setting thresholds, etc.
	Storm surge incidence	d y^{-1}	Meteorological agencies	A	*	*		R	
	Significant wave height	m	Meteorological agencies	A	*	*		R	

Group	Variable	Metrics	Measurement procedure	f	Intertidal[1]	Coastal[1]	Non-coastal[1]	App	Observations*
Sediment dynamics	**Sedimentation/ erosion patterns**	mm y^{-1}	Sedimentation-erosion table (SET)	A	*	*	*	E	Can be combined with the feldspar horizon technique.
		gDW m^{-2} y^{-1}	Sediment traps	S	*	*	*	R	
	Sediment sources	gDW l^{-1}	Turbidimeters	S	*	*		R	This variable changes daily and seasonally, but may be decisive for the resilience of tidal and coastal salt marshes.
1.2 Chemical characteristics									
Salinity and nutrient conditions	**Soil salinity**	ppm	Plot surveys	S	*	*	*	E	
	Nutrient content in soil (C, N, P, SO4, etc)	ppm	Plot surveys	S	*	*	*	E/R	This variable is essential if major changes in diversity or coverage are detected; otherwise, it is recommended.
Organic matter content	**Organic matter in soil**	%	Plot surveys	S	*	*	*	E/R	Organic matter is a useful proxy for anoxia and carbon storage. Its measurement is essential when notable changes in species diversity or vegetation cover occur; otherwise, it is recommended.
2. Biotic characteristics									
2.1 Compositional characteristics									
Species assemblage	**Characteristic plant species richness**	Number of species per unit area	Plot surveys	S	*	*	*	E	Lists of characteristic species should be defined according to habitat type and region.
	Characteristic animal species richness	Number of species per unit area	Plot surveys	S	*	*	*	E	Lists of characteristic species should be defined according to habitat type and region.
	Non characteristic plant species richness	Number of species per unit area	Plot surveys	S	*	*	*	E	The presence of non-characteristic species indicates changes in abiotic conditions and a potential shift or degradation of the habitat.
	Presence of harmful invasive species (HIS)	Presence recorded	Plot surveys	E	*	*	*	E	A list of potential HIS and IAS should be established according to habitat type and region.

Group	Variable	Metrics	Measurement procedure	f	Intertidal[1]	Coastal[1]	Non-coastal[1]	App	Observations*
2.2 Structural characteristics									
Community structure	**Canopy height**	m	Plot surveys	S	*	*	*	E	Changes in canopy height may reflect variations in physicochemical conditions or the presence of invasive species.
	Species coverage	%	Plot surveys	S	*	*	*	E	Percentage of the plot covered by the dominant characteristic species
	Zonation patterns	Width, elevation, and number of zones	Transect surveys across the ecosystem; dGPS	S	*	*	*	E	Crucial for tidal and coastal systems, as it supports habitat coexistence and natural successional stages. n intertidal systems, habitat 1140 forms part of the natural zonation pattern and should be noted when present.
	Erosion cliffs	Cliff height and length	Plot surveys; dGPS and GIS	S	*			R	Once triggered, marsh cliff erosion continues until foreshore expansion permits pioneer marsh recovery (Bouma et al., 2016).
2.3 Functional characteristics									
Functionality	**Connectivity / naturalness**	Matrix with 3 categories	GIS-based visual inspection	6Y	*	*	*	E	Presence or absence of anthropogenic structures that alter creek morphology and/or water movement.
3. Landscape/seascape characteristics									
3.1 Habitat spatial scale and connectivity									
	Extent area	ha	Remote sensing	6Y	*	*	*	E	
	Fragmentation / patchiness	Number, size, and distance between patches	Remote sensing	6Y	*	*	*	E	Fragmentation of continuous habitats is negative; however, an increase in patches within pioneer areas is considered positive.
	Slope	%	dGPS or centimetric DTM transects	6Y	*	*		R	A gentle slope is favourable for the development and persistence of extensive salt marshes.
	Accommodation area	ha, %	Remote sensing	6Y	*	*	*	E	Available area (ha) and the proportion of this area suitable for habitat displacement to similar relative elevations to MSL, based on a 50-year worst-case SLR scenario and assuming the current habitat slope.

3.2 Guidelines for the establishment of reference and threshold values, and obtaining condition indicators for the variables measured

Establishing reference values and thresholds is essential for determining whether habitats are in good condition or have become degraded. Reference values represent the desired state of an ecosystem, typically reflecting intact or minimally disturbed conditions. These values serve as benchmarks for assessing habitat condition.

A reference level is the value of a variable under reference conditions, against which it is meaningful to compare past, present or future measurements. The difference between a variable's measured value and its reference level represents its distance from the reference condition.

Reference levels are typically defined with upper and lower values reflecting the endpoints of a condition variable's range, which can then be used in re-scaling. For instance, the highest value may represent a natural state, while the lowest value may represent a degraded state where ecosystem processes fall below the threshold required to maintain function (Keith et al., 2013, in United Nations et. al., 2021). For example, pH values in freshwater ecosystems clearly indicate whether biological life can be sustained, while soil nutrient enrichment beyond a certain threshold can lead to the loss of sensitive species.

The thresholds that define the values of salt marsh and salt meadow variables for favourable conditions vary according to the specific variable, habitat, and biogeographical gradients. These harmonization guidelines do not aim to provide specific threshold values but rather to establish the primary criteria for setting reference values. These reference values should determine the habitat condition of salt marsh and salt meadow habitats, considering the ecological variability of the habitat across its range.

For the harmonization of reference values, a set of common requirements should be considered, including:

- For a given habitat, the final assessment of its condition - based on the reference values and thresholds of the variables characterising the habitat - should be consistent across different MSs, after accounting for the contextual factors specific to each MS (e.g., climate).
- Thresholds must account for the natural variability of habitats across their range. Consequently, different threshold or reference values for the same habitat type may be appropriate in different MSs or in different regions within a single MS.
- Establishing reference values requires information external to the evaluated site, which can provide insight into the condition of the habitat and be translated into variable values that characterise that condition.
- Reference values should meet criteria of validity (ecological relevance), robustness (reliability), transparency, and applicability (Czúcz et al., 2021b; Jakobsson et al., 2020).
- Thresholds, limits, and reference values should be tested using sufficiently robust datasets that represent the full range of habitat conditions, from degraded to high-quality sites.
- Each MS should provide a clear, justified, and comprehensible description of the methodology used to establish threshold and reference values for each variable
- The methodologies should be designed for regular evaluation and improvement, based on the best available scientific knowledge. Any modifications made – and their implications for past monitoring data – must be communicated transparently.
- Each MS should provide a clear, justified, and comprehensible description of the methodology used to establish reference and threshold values for each variable.

- The methodologies should be designed for regular evaluation and improvement, based on the best available scientific knowledge. Any modifications made – and their implications for past monitoring data – must be communicated transparently.
- Joint training or guidance on setting reference values and thresholds should be offered to experts from the different MSs in order to achieve ensure harmonised approaches.

Reference values should be based on empirical data from relevant reference systems or historical reference periods (Jakobsson et al., 2020). However, in practice, such baseline data for salt marshes and salt meadows are scarce, incomplete, or non-existent. Based on Jakobsson et al. (2020), five main approaches for setting reference values could be applied to salt marshes and salt meadows, including: (1) absolute biophysical boundaries, (2) reference areas, (3) modelling, (4) statistical assessments and (5) expert judgement.

Absolute biophysical boundaries

These refer to situations in which observed values of variables exceed the typical physical and chemical limits (e.g., inundation frequency, soil salinity) or biotic limits (e.g., presence of harmful invasive species) that define the habitat. When such limits are exceeded, the habitat cannot be in good condition. These thresholds therefore indicate negative impacts on the favourable condition of the habitat.

- Advantages: This approach establishes provides robust and transparent criteria robust that are that are clearly linked to the ecological integrity of the habitat.
- Disadvantages: It is applicable to a limited number of variables, typically those with direct negative impacts on habitat condition.

Reference areas

This approach is based on identifying areas or communities considered to be in good condition (Stoddard 2006, Jakobsson 2020, Keith et al., 2020). These serve as reference cases from which the reference values can be derived. Therefore, their careful selection – and the availability of a sufficient number of such cases – is essential for ensuring the reliability of the reference value estimates (Soranno et al., 2011). While this method may appear straightforward, it is often limited by the scarcity of suitable sites, especially in landscapes that have been historically modified.

- Advantages: Providing that sufficient data from high-quality cases are available, this approach offers empirical validity and reliability by directly linking variable values to habitat condition.
- Disadvantages: Methodological challenges arise due to the difficulty of identifying a sufficient number of suitable reference sites in historically altered environments.

Modelling the relationships between variables and condition

This approach assumes a relationship between variable values and habitat condition. When determining threshold and reference values, models that describe these relationships share a conceptual basis with methodologies based on dose-response curves. Such models assume that certain cases of good condition correlate with specific levels of a condition variable.

The advantage of modelling is that it allows reference values to be inferred where empirical examples of good condition or undisturbed condition are lacking. In these situations, information from known empirical examples can be extrapolated to other contexts, such as locations along a climatic gradient.

Various modelling procedures are available. Functional relationships – linear, saturated, or humped – can be applied (Stoddard et al. 2006, Jakobsson et al. 2020). Correlative climate niche models can also be used to estimate the suitability of species sets (i.e., variables that characterise the habitat) at different points along the climatic gradient (Jakobsson et al. 2020).

Modelling approaches require the availability of field databases and the development and validation of the corresponding models to allow calculation of reference values. However, this approach allows to use data from historical records and/or selected reference sites as well as empirical models (Stoddard et al., 2006).

- Advantages: modelling approaches are flexible, transparent and encompass a variety of different procedures based on functional relationships between variables and condition (validity), drawing on scientific knowledge from multiple disciplines. They can also be applied to obtain reference values when empirical examples of good or undisturbed condition are lacking.
- Disadvantages: the information available to build models is insufficient or unreliable for many variables. Outputs are highly sensitive to the chosen modelling procedure and underlying assumptions, and expert judgement is ultimately required at multiple stages of the modelling process.

Statistical assessments

This approach is based on quantitative data from databases, such as habitat inventories, which report the distribution of variables within a given habitat. It assumes that higher values of certain variables correspond to good condition when a positive relationship exists, and vice versa. For such variables, high percentile values or confidence intervals (e.g., 95%, Jakobsson et al., 2020), or differences from the maximum observed values (Storch et al., 2018), may be used.

For variables with a negative impact on habitat condition, low (e.g., 5%) or minimum values are applied, while for variables that show a humped-shaped (non-linear) relationship with condition - peaking at intermediate values (e.g., gap occurrence, browsing) - a combination of high and low percentiles may be used.

Unfortunately, nowadays, there are scarce inventories of salt marshes and salt meadows with habitat characterization further than location and extension.

- Advantages: this approach can be applied with reasonable ease by users with statistical training. It is transparent, replicable and minimally subjective.
- Disadvantages: the existence of appropriate, quantitative datasets representing the reference state is essential for this method. Its reliability depends on the distribution of the condition classes (from bad to good) in the dataset and how well this distribution corresponds to empirical situations of good condition. As a result, it may lead to under- or over-estimation of good condition and may be less reliable for habitats that are poorly represented in the dataset.

Expert judgement

Setting of reference values and thresholds based expert judgement is common practice, particularly where other sources of information are lacking – for instance, in certain non-abundant habitats where experts have developed empirical knowledge of habitat conditions. However, this approach is often criticised for its limited transparency and the level of expertise may be insufficient in some cases. For this reason, it is sometimes considered a last-resort option for many variables.

Nonetheless, for certain variables such as species richness or presence and extent of and harmful invasive species (Table 11), expert judgment may be appropriate for establishing thresholds and reference values. In other cases, it can also serve as a complement to other approaches.

In all situation, it is advisable, to apply expert judgement through protocols based on consensus and consultation with multiple experts of comparable experience. This should include clear (e.g., standardised questionnaires) and transparent documentation of how conclusions were reached (Stoddard et al., 2006). A further limitation is the lack of available experts for certain habitats, which can hamper the correct implementation of this approach.

- Advantages: This approach is easy to apply and is commonly used.
- Disadvantages: It entails a high degree of subjectivity and low transparency, which limits replicability and reliability. Its use may also be constrained by the scarcity of suitable experts for particular habitats and Member States.

Table 11 provides an overview of the approaches used to establish reference values and thresholds for the proposed condition variables intended for harmonization. These approaches are not mutually exclusive and are often applied in combination. For example, expert judgment is required when defining reference areas for good condition or when making modelling decisions about the relationship between variables and condition. Similarly, modelling-based approaches can complement those based on reference areas, and may also be integrated with statistical methods.

Habitat condition assessments are based on determining whether the variables used indicate good or not good condition. However, it is common practice to define more than two categories for each variable, e.g., good, medium, bad - as observed in the analysis of methodologies used by Member States. The criteria for assigning these condition categories vary depending on the characteristics of each variable. For example, categorical variables may involve thresholds such as “no alien species allowed”, while quantitative variables may follow linear or non-linear relationships with condition (Jakobsson et al. 2020).

Table 11. Approaches to establish thresholds and reference values for determining good condition, applicable to the proposed variables for salt marshes and salt meadows

The intensity of the colour indicates the priority of the approach, with dark grey indicating the preferred or commonly applied criteria.

Variable	Absolute biophysical boundaries	Reference areas	Modelling	Statistical assess-ments	Expert judgement
Elevation relative to LAT					
Tidal range					
Spring groundwater level					
Rainfall					
Storm surge incidence					
Wave energy					
Sedimentation/erosion patterns					
Sediment sources					
Soil salinity					
Nutrient content in soil (C, N, P, SO_4, etc.)					
Presence and extent of characteristic species					
Species richness					
Presence and extent of Harmful Invasive Species (HIS)					
Canopy height					
Species distribution and coverage					
Zonation patterns					
Erosion cliffs					
Connectivity / naturalness					
Extent (area)					
Fragmentation / patchiness					
Slope					

This classification of variable values – whether quantitative or categorical – into condition categories (i.e., good and not good; or good, medium and bad) corresponds to the scaling process for joint evaluation through aggregation procedures, as described in the following section. Condition categories can be translated into numerical values (e.g., good = 2, medium = 1, bad = 0). Alternatively, where quantitative values for the variables are available, these can be directly standardised for use in aggregation.

In habitat condition assessments, each characteristic and its associated variable is likely to be measured in a different unit. Owing to the different metrics and magnitudes used for the variables that characterise habitats, the values obtained from their measurement requite som form of standardisation – e.g., through rescaling – in order build indicators that combine multiple variables. These values are normalised using reference levels and reference conditions, allowing comparison across variables. Measurement values are thus scaled in relation to the reference levels, thereby normalised to a common scale and aligned direction of change. They can then be combined to form a composite index or used to obtain an overall condition result through appropriate aggregation approaches (see further details in Section 3.3 on Aggregation.

3.3 Guidelines for the aggregation of variables at the local level

Ecological assessments require the integration of physical, chemical, and biological quality elements. The choice of the aggregation method for combining these partial assessments into an overall evaluation has been widely discussed within the scientific community, as it can significantly influence the final outcome. Various approaches can be used to integrate the values of measured variables into an overall index reflecting the condition of habitat types at the local scale (e.g., monitoring plot, station or site).

Applying appropriate aggregation approaches is essential for categorising condition at the local scale as good or not good, since the proportions of habitat type area in good/not good condition is the key information needed for evaluating the conservation status of structure and functions at the biogeographical level.

3.3.1 Overview of aggregation methods

Based on the literature (e.g., Langhans et al., 2014; Borja et al., 2014), two main aggregation approaches can be identified: the one-out, all-out rule (minimum aggregation) and additive aggregation (e.g., simple addition, arithmetic mean, geometric mean, etc.).

Further information on aggregation approaches and methods is provided below.

Minimum aggregation, or the one-out, all-out rule

For the minimum aggregation, the aggregated value is calculated as the minimum of the values of the measured variables.

The one-out, all-out (OOAO) rule has been recommended for assessing ecological status under the Water Framework Directive (CIS, 2003). The principle behind this minimum aggregation method is that a water body cannot be classified as having good ecological status if any of the measured quality elements fail to meet the required threshold. This is considered a precautionary and rigorous approach, but it has also been criticised for potentially underestimating the true overall status.

A precautionary OOAO approach is also used in the aggregation of parameters when assessing conservation status under the Habitats Directives, the IUCN Red List of Species and the IUCN Red List of Ecosystems.

Conditional rules

Conditional rules require that a certain proportion of variables meet their respective thresholds in order for the overall assessment to achieve a good condition rating. For example, the overall status may be considered as not good when a specific number of variables fail to meet their thresholds.

Simple additive methods and averaging approaches

Simple additive methods calculate an aggregated value as the sum of the n values (vi) of the variables.

Averaging approaches are among the most commonly used methods for aggregating indicators. These include straightforward calculations such as the arithmetic mean, weighted average, median, or combinations thereof, to produce an overall assessment value.

Weighting

Differential weighting of indicators may be applied when calculating sums, means, or medians. The choice of weighting system should reflect the relative importance of each indicator in determining the overall condition of the ecosystem. Ideally, the approach should be supported by a clear scientific rationale and informed by input from ecologists with expertise in the relevant ecosystem types.

However, a robust basis for assigning weights is not always available. In such cases, weighting often relies on expert judgment, which can be subjective, as expert opinions may differ considerably.

Normalization of variables values (rescaling)

In the assessment of habitat condition, each characteristic and associated variable is likely to involve the use of different measurement units. To ensure comparability, the measured values of variables are often normalised to a common scale (e.g., 0 to 1 or 0 to 100). This involves rescaling the raw data based on reference values or thresholds that define the boundary between good and not good condition for each variable. By rescaling the condition variables, indicators are standardised to the same scale, making it possible to aggregate them into condition indices that reflect the overall condition at a given plot or location.

Figure 6. Example of deriving condition indicators by rescaling the values obtained for variables, based on upper and lower reference levels

$$Condition\ indicator = \frac{(V-VL)}{(VH-VL)}$$

Where:

- V is the measured/observed value of the variable,
- VH is the high condition value for the variable (upper reference level),
- VL is the low condition value (lower reference level).

Source: Vallecillo et al. (2022)

3.3.2 Proposal for the aggregation of measured variables

A quantitative aggregation method should be applied to integrate all essential and specific variables measured to assess the habitat condition. The method should be applied consistently across the habitat range in order to obtain comparable results. The main steps for aggregation are described below.

Step 1 – Normalisation of the variables

The quantitative values obtained for each variable should be normalised by rescaling based on reference values (as described above). The value of each variable will be thus in the range from 0 to 1.

Step 2 – Aggregation of normalised variables

The aggregated value is then calculated by the aggregation of the normalised values of the variables. For the sake of simplicity, and considering the difficulties to suggest a more complex method or index, we describe here a preliminary proposal for aggregation based on the arithmetic mean with normalisation of the values obtained for each of the measured variables, which could be used to determine the habitat condition at the local scale, as summarised in the following equation:

$$Local\ condition = \sum_{1}^{n} v_i/\mathrm{n}$$

Where n is the number of variables, v_i the rescaled value of the corresponding variable (between 0 and 1). The aggregated value would range between 0 and 1.

An alternative method would be to use the weighted average, in which the weight of each variable should be decided, justified and agreed upon for each habitat type by all the MSs that would apply the method. This method can be formulated with the following equation:

$$Local\ condition = \sum_{1}^{n} v_i * w_i/\mathrm{n}$$

Where n is the number of variables, v_i the rescaled value of the corresponding variable (between 0 and 1) and w_i the corresponding weight, with $\sum w_i = 1$. The aggregated value would range between 0 and 1.

This second method, however, presents some difficulties when assigning weights to the variables, which must be based on a proper evaluation of their importance and influence on the habitat condition, based on a robust scientific knowledge. It also requires reaching a consensus on the weights assigned to the variables measured for each type of habitat, among all the countries that must assess its condition. This is a crucial aspect to obtain comparable results in the assessments carried out by all the Member States.

Step 3 – Identify the threshold to determine good/not good condition at the local scale

Finally, a threshold must be applied to the aggregated value to distinguish between good and not good overall condition. This is a crucial step and, wherever possible, this threshold should be established based on empirical data from reference localities in good condition and from localities showing a degraded state. Where such reference localities are not fully available, modelling to obtain such thresholds could be applied.

Limit between good/not good condition

3.4 Guidelines for aggregation at the biogeographical region scale

As a minimum requirement, Member States must follow the recommendations set out in the Article 17 reporting guidelines for the 2019-2024 period (European Commission, 2023). These guidelines recommend a threshold of 90% of the habitat type's area being in good condition in order for the 'structure and functions' parameter to be assessed as favourable. Ideally, the entire habitat area should be in good condition for this parameter to be considered favourable. However, achieving this in practice is often difficult. It may therefore be acceptable for a limited portion of the habitat type to be in not-good condition, while still assessing the 'structure and functions' parameter as favourable.

This indicative threshold may be adjusted based on the rarity or spatial extent of the habitat type. For example, in the case of rare habitats covering only a few tens of square kilometres within a biogeographical region, the threshold may approach 100% (European Commission, 2023).

Conversely, if more than 25% of the habitat area is reported as not in good condition, the 'structure and functions' parameter must be assessed as unfavourable-bad.

This rule highlights the importance of a sampling design that adequately represents the total area and ecological diversity of the habitat type (see further guidance In Section 3.5.1).

3.5 Guidelines on general sampling methods and protocols

Harmonised monitoring protocols are essential for assessing habitat conditions across Europe. They should provide standardised methods for data collection, analysis, and interpretation to ensure consistency and comparability over time and between regions. This section outlines key recommendations on sampling design and monitoring protocols.

3.5.1 Sampling protocol

Salt marshes can contain more than one habitat type (Figure 1), due in part to the influence of inundation and salinity patterns on biodiversity establishment (Contreras-Cruzado et al., 2017). This results in zonation patterns that develop parallel to the water line, where each band corresponds to a different habitat type as defined by the Habitats Directive classification. Therefore, it is recommended to use transect-based sampling designs – or modifications thereof – that allow assessment of habitat condition at two scales: within each zonation band (habitat scale) and across the full environmental gradient (landscape scale) (see Figure 7).

To characterise salt marshes and salt meadows, a sampling design is suggested that includes six plots per homogeneous vegetation band (i.e., per HD habitat type), evenly distributed within each habitat. Where feasible, plots should be spaced at least 20 metres apart. The design should include as many bands as there are habitat types present within the salt marsh or salt meadow. The recommended approach resembles a transect design but incorporates six replicates per habitat type (Figure 7).

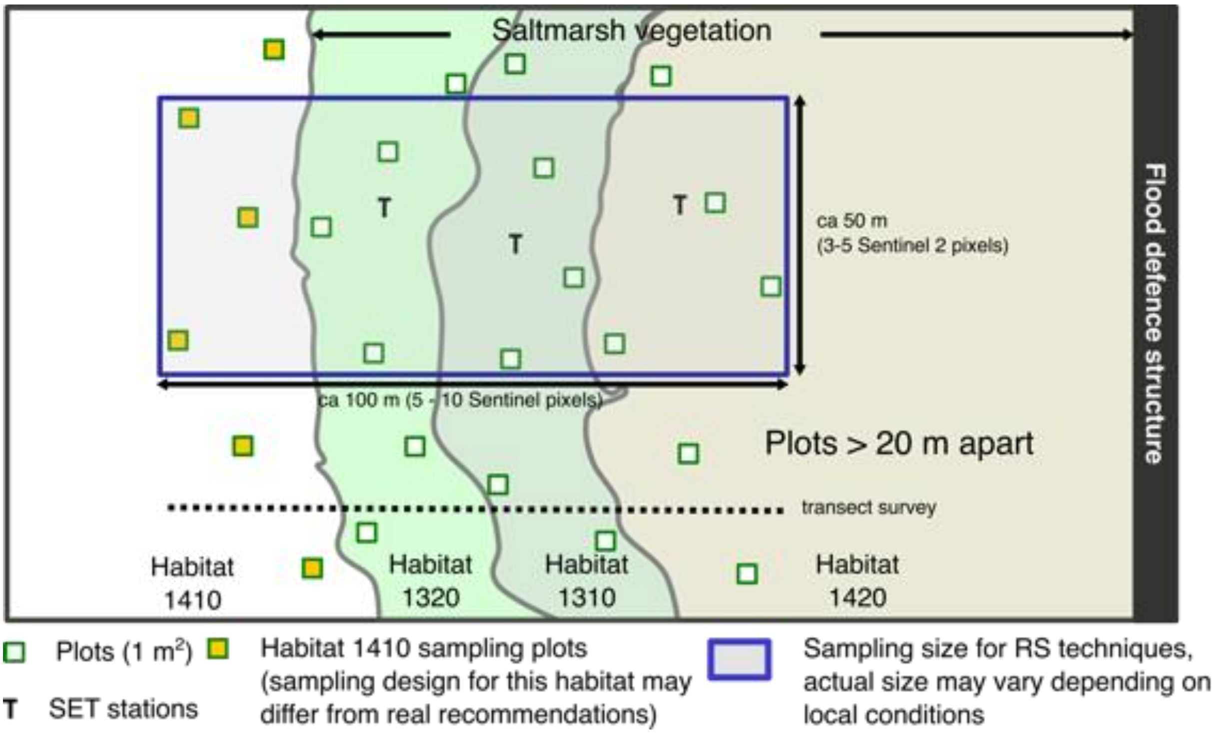
For homogenous vegetation, a plot size of 1 m^2 is recommended. However, if the vegetation distribution is patchy and cannot be adequately characterised using a 1 m^2 plot, circular plots with a 2.5 m radius, or square plots of equivalent area, are recommended.

Elevation and soil salinity should be measured within each plot, using harmonised methods across Member States. For soil salinity, suitable methods include electrical conductivity or salt weight proportions. If different methods are used across countries, intercalibration exercises among MSs are recommended to ensure comparability.

Elevation should be characterised not only at the plot scale, but also at the landscape scale. At the landscape level, it is advisable to generate a high-resolution DTM raster (e.g., with centimetric precision), or at minimum, to conduct transversal transects using dGPS technology.

An example of this type of sampling design for characterising salt marshes was applied in the FAST project (de Vries et al., 2018). In this project, the sampling design was tailored to provide information at two spatial scales: a small scale with high spatial resolution for the characterisation of each zonation band (six plots per band), and a large scale with low spatial resolution (i.e., 50 x 50 m) to calibrate Landsat imagery and characterise the system at the landscape level.

Figure 7. Schematic example of the proposed sampling design for assessing salt marsh habitat condition



Characterisation of individual habitats corresponds to the habitat scale, while characterisation across the full salt marsh gradient corresponds to the landscape scale.
Source: Gloria Peralta

To accommodate both scales, sampling plots were spaced at distances greater than 45 m from each other. Today, Sentinel-2 imagery offers higher spatial resolution (up to 10 x 10 m), allowing for alternative spatial configurations of sampling plots where appropriate. Nevertheless, UAV technology is likely to remain necessary to effectively bridge the gap between plot-scale and satellite-scale observations (see section 2.7 for further details).

3.5.2 Monitoring frequency

Article 17 of the Habitats Directive requires that habitat condition be assessed within a maximum period of six years. However, this six-year period may be fulfilled through various approaches, depending on the resources available to each Member State. Not all plots or variables need to be measured every six years.

For plot-based monitoring, MS may designate a large number of monitoring sites, with only a subset surveyed each season. This approach allows for the progressive collection of a representative dataset, ensuring that a sufficient number of plots are fully monitored at least once within each six-year reporting cycle.

Within this period, it is important to consider that salt marshes and salt meadows exhibit strong seasonality (Curcio et al., 2024). As such, seasonal sampling is usually recommended for the biotic characterisation of habitats (see Table 10). Alternatively, annual sampling may be used, provided that measurements are taken consistently at the same time each year, preferably during the peak biomass period. In such cases, it is recommended that the sampling window be agreed among MSs, to ensure data comparability across countries.

The establishment of a harmonised European database for salt marsh and salt meadow monitoring data is strongly encouraged. This would support coordinated evaluation of these habitats at the European level.

3.6 Selecting monitoring localities and sampling design

The selection of localities for monitoring salt marshes and salt meadows must ensure adequate representation of ecological and geographical variability across their distribution. These habitats occur in multiple biogeographical regions (e.g., Atlantic, Mediterranean, Continental) and marine sub-bioregions (e.g., Bay of Biscay, Gulf of Cádiz, Western Mediterranean). In the case of tidal and coastal systems, classification and sampling based on marine sub-bioregions is often more appropriate, as it better reflects differences in hydrodynamics, tidal regimes, and salinity patterns. Monitoring sites should be spatially distributed across the full extent of the habitat type to avoid geographical bias and to adequately capture regional and environmental heterogeneity (Figure 8). Special attention should be given to the coordination of salt marsh and salt meadow monitoring with adjacent and ecologically connected habitats (e.g., estuaries, intertidal mudflats, and coastal dune systems) in order to integrate assessments of ecological functioning at the landscape level.

The selection of monitoring localities should be grounded in a comprehensive inventory, supported by detailed cartography and ecological characterization of habitat types and their variability. Both the choice of localities and the definition of sampling effort, including the number of plots and the required statistical power, are critical to ensure that the results are robust, comparable, and representative at the biogeographical scale.

Based on this foundational information, the identification and selection of sampling sites must follow a systematic and well-informed approach. The following criteria should guide the selection of monitoring localities to ensure representativeness and robustness in the assessment of salt marshes and salt meadows.

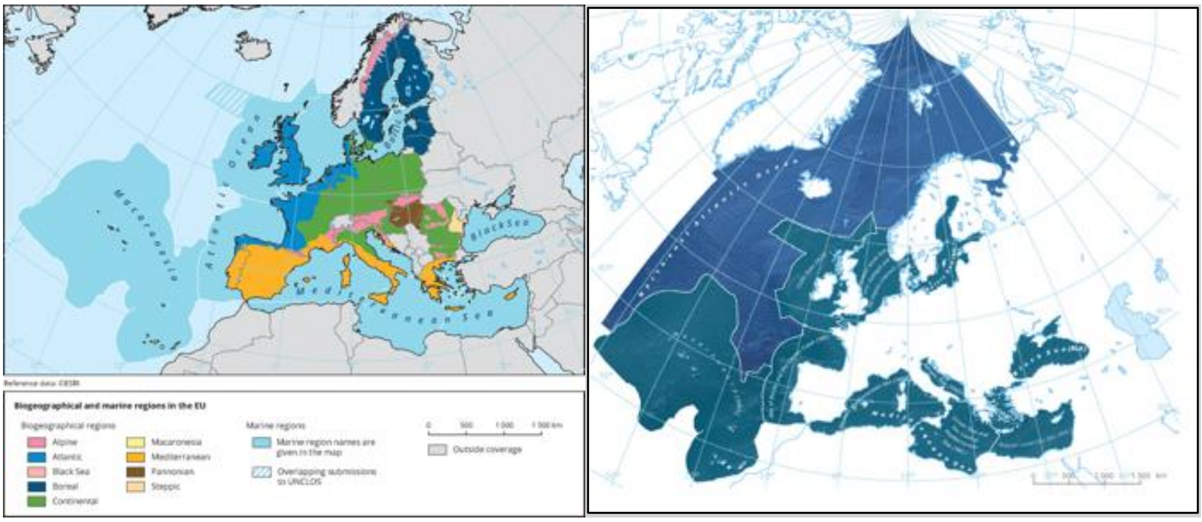
Criteria for selecting monitoring localities

- **Ecological variability:** Localities must represent the full range of ecological diversity and variability within the habitat type. Selection should include different ecotypes or subtypes, successional stages, and reflect key environmental gradients (e.g., salinity, soil moisture, altitude, topography, and geomorphology).
- **Spatial coverage:** Adequate spatial coverage is essential to capture habitat heterogeneity. Localities should be selected across the full geographical range of the habitat

type within the region, ensuring they are well distributed and represent a significant proportion of the habitat's total occupied area.

- **Degree of conservation and exposure to pressures and threats:** Sites should span a gradient of conservation conditions, from well-preserved to degraded areas. Including areas subject to different pressures (e.g., urbanisation, agriculture, climate change) ensures that the monitoring captures the range of pressures and threats affecting habitat condition.
- **Presence inside and outside Natura 2000:** Monitoring must take place both within and outside Natura 2000 sites. The number and distribution of plots should be proportional to the habitat's occurrence inside and outside the network, enabling an integrated conservation status assessment.
- **Landscape-scale fragmentation:** The selection should reflect differences in landscape configuration, considering metrics such as patch size and connectivity. Including both isolated and well-connected sites allow evaluation of fragmentation effects on habitat integrity.
- **Data gaps:** Priority should be given to areas with limited or no previous information. Including historically under-sampled regions helps to complete the knowledge base and reduce geographical bias in habitat assessments.
- **Accessibility and operational feasibility:** Sites must be logistically viable for regular monitoring. Factors such as proximity to access roads, terrain, safety, and the ability to transport equipment should be considered to ensure practical implementation.
- **Historical data and existing monitoring sites:** Where available, existing monitoring sites with historical datasets should be incorporated to enable assessment of long-term changes and trends in habitat condition. These sites provide.

Figure 8. Biogeographical and marine regions (top) and marine subregions (bottom) for EU-28 countries



Source: European Environment Agency (EEA).

Once the sampling localities have been selected for each habitat type, the minimum number of plots per locality - and across the biogeographical region - must be calculated to balance sampling effort with the need for representative data.

The number of sampling plots influences two statistical properties (1) the precision of the estimated indicators, and (2) the statistical power to detect meaningful changes or trends. The number of plots must be statistically sufficient to detect changes and trends with the desired level of confidence. Appropriate statistical methods should be applied to determine and adequate sample size.

Considering the heterogeneity of habitat types, it is highly recommended to consult with a sampling statistician when determining sample size, i.e., the minimum number of plots required to ensure representativity and statistical significance.

Some key elements for ensuring a representation of habitat condition in the sample are summarised below.

Key elements for statistical representation

- **Sample Size and Distribution:** The number of sampling localities and plots must be sufficient to provide a statistically robust sample size. This ensures that the collected data can be generalised to the habitat type within the biogeographical region. Statistical methods such as stratified random sampling are often applied to ensure that all habitat subtypes and environmental gradients are adequately represented.
- **Sampling Design:** Within each sampling area or locality, multiple plots or transects are established to collect detailed data on vegetation, soil, and other relevant ecological variables. The number and distribution of plots depend on the size of the habitat patch and its internal variability. Sampling areas (e.g., plots, transects) should be laid out with consideration of the main ecological gradients, such as altitude, moisture, and exposure to sea influence.
- **Replication and Randomization:** Replicating sampling units within each locality and randomising the location of sampling plots help reduce bias and increase the reliability of the data. Randomised plot locations also ensure that sampling captures the natural variability within the habitat.

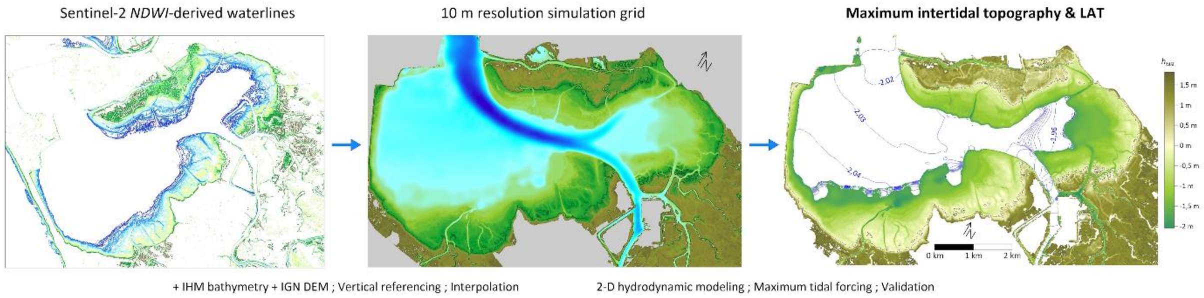
3.7 Use of available data sources, open data bases, new technologies and modelling

Remote sensing techniques have become invaluable tools for acquiring spatial environmental information and enabling the monitoring of large areas with consistent temporal resolution (Macintyre et al., 2020). Traditional platforms such as satellites and aerial systems have been widely used in regional-scale studies, including the mapping tidal marshes (Byrd et al., 2018), the evaluation of natural disasters (Caballero et al., 2019), the monitoring of vegetation cover (Gandhi et al., 2015), and the management of coastal areas (Ouellette & Getinet, 2016).

In particular, optical satellite remote sensing provides critical data for understanding historical changes in salt marsh extent and for supporting coastal decision-making. These satellites collect reflectance data across the visible and infrared spectrum, which can be used to calculate spectral indices (SIs). For instance, water indices combined with hydrodynamic modelling have been used successfully to map digital elevation models (DEMs) in intertidal areas (González et al., 2023; see Figure 9).

When applied to vegetation, these indices are referred to as vegetation indices (VIs). In salt marsh environments, satellite-derived VIs has been used to distinguish between different vegetation types (Sun et al., 2020), analyse global salt marsh changes and associated carbon emissions (Campbell et al., 2022), map salt marsh extents (Lopes et al., 2020), and produce multi-temporal classification maps of salt marsh communities using monthly NDVI time series (Sun et al., 2018).

Figure 9. Graphical abstract of the methodology used to generate high-resolution intertidal digital terrain models (DTMs) from Sentinel-2 imagery and hydrodynamic modelling



Source: Carlos J. González Mejías (IHM, Spain).

Despite their broad applicability, freely available satellite products are limited by their spectral, spatial, and temporal resolution, which can hinder the detailed modelling of ecological processes (Hossain et al., 2015; Nex & Remondino, 2014). Unmanned aerial vehicles (UAVs) help bridge this gap by offering high-resolution, rapid, and cost-effective monitoring methods. UAVs can be equipped with a variety of sensors, including high-resolution photogrammetry cameras as well as more advanced technologies such as thermographic, multispectral, LiDAR, and hyperspectral sensors.

Three remote sensing techniques show particular promise for high-quality monitoring of salt marshes: Photogrammetry, which produces topographic products through Structure-from-Motion (SfM) (Westoby et al., 2012); Light Detection and Ranging (LiDAR), which generates accurate 3D point clouds for high-resolution topography and digital elevation model (DEM) creation (Brock & Purkis, 2009); and Multispectral imaging, which provides essential data for vegetation mapping.

The combination of multispectral and LiDAR sensors mounted on UAVs has yielded excellent results for assessing extent, vegetation cover, and canopy height of halophytes in intertidal environments at the landscape scale (Figure 8; Curcio et al., 2024). In addition, hyperspectral sensors can distinguish between different plant species at specific sites (Curcio et al., 2023).

When combined with satellite-derived vegetation presence/absence data, accurate DTM, and sea level rise (SLR) modelling, UAV-based products offer significant opportunities for monitoring habitat conditions and tracking temporal changes in salt marshes and salt meadows.

4 Guidelines to assess fragmentation at appropriate scales

4.1 Effects of fragmentation on saltmarshes and salt meadows

Patchiness and fragmentation are key ecological concepts used to describe spatial patterns and processes within habitats. While both relate to the distribution and arrangement of habitat areas, they have distinct meanings and implications (see Table 12).

Patchiness refers to the uneven distribution of resources, organisms, or habitat types within an ecosystem. It is a natural characteristic of most environments, where elements are not uniformly distributed but instead form clusters – patches – separated by a matrix of different habitat types (Macdonald & Johnson, 2015). In coastal salt marshes, such spatial heterogeneity plays a crucial role in establishing bio-geomorphic feedbacks, which are highly relevant for the functioning of these ecosystems (Van Wesenbeeck et al., 2008).

Fragmentation, in contrast, is defined as the process by which a large, continuous habitat is broken into smaller, isolated patches (Hagen-Kissling et al., 2012). Fragmentation can result from natural events or human activities such as urbanisation, deforestation, or agricultural expansion (Martin et al., 2021). This process leads to habitat loss, increased edge effects, and the isolation of populations – factors that can threaten both biodiversity and ecosystem functioning.

In the context of salt marshes and salt meadows, fragmentation may severely impact biodiversity and the provision of key ecosystem services. It can lead to the degradation of functions such as coastal hazard protection (Zhang et al., 2023) and blue carbon storage (Rippel et al., 2020). Therefore, the evaluation of habitat patchiness should be integrated into assessments of habitat condition. Crucially, determining whether observed patch patterns reflect natural variability or result from fragmentation processes requires temporal analysis of spatial patterns over time.

In summary, the assessment of fragmentation requires repeated spatial characterisation of habitat patchiness over time. This can be based on updated GIS-derived habitat maps, which MSs are expected to provide every six years under their obligations for Article 17 of the Habitats Directive (Ellwanger et al., 2018).

Table 12. Comparison of characteristics and ecological consequences of fragmentation and natural patchiness

Feature	Fragmentation	Natural patchiness
Cause	Human activities or large-scale disturbances	Natural environmental factors (e.g. soil, hydrology, climate)
Impact on habitat structure	Leads to habitat loss and isolation	Maintains ecological heterogeneity and niche availability
Effect on species	Reduces connectivity; limits movement and gene flow	Supports biodiversity and natural ecological processes
Habitat Size	Often results in smaller, isolated patches	Patches may vary in size but are naturally occurring
Ecological function	Can degrade ecosystem functions (e.g. coastal protection, carbon storage)	Typically supports ecosystem functions through spatial heterogeneity

Alternatively, fragmentation assessment can be supported by image analysis, such as the interpretation of satellite or drone imagery, or orthophotography. Characterising fragmentation involves the use of spatial landscape metrics and the quantification of changes in these metrics over time – known as dynamic characterisation. The degree and trend of habitat fragmentation can be evaluated through the analysis of landscape metrics applied to geomorphological diachronic maps, produced by comparing habitat cartography across different time periods.

4.2 Landscape metrics

In fragmented landscapes, spatial metrics focus is used to characterise the patchiness of the system. A patch is the smallest spatial unit in this context, defined as a homogeneous area (e.g., a single polygon in GIS) that differs from its surrounding environment (Forman, 1995).

To assess the degree and trend of the fragmentation, the dynamics of landscape patchiness must be analysed in each change map (Aranda et al., 2022). The metrics used to describe patchiness should encompass multiple dimensions of patch distribution, including area, shape, aggregation, and diversity (Ebenezer & Kavitha, 2021). Together, these metrics provide a comprehensive assessment of landscape fragmentation by evaluating key spatial properties:

- **Total habitat area:** Total area contained within the boundary of the habitat.
- **Total patch area:** Total area occupied by patches within the boundary of the habitat.
- **Number of patches:** Number of patches within the boundary of the studied habitat.
- **Total patch perimeter:** Sum of the perimeters of all the patches.
- **Patch size:** Area and perimeter of individual patches.
- **Perimeter-area ratio:** The ratio of the perimeter to the area of a patch.
- **Landscape Shape Index (LSI):** A metric to measure the complexity of the landscape configuration by comparing the total perimeter of all patches with the minimum perimeter for the same number of patches. Values greater than 1 (LSI>1) indicate more complex and irregular patch shapes, reflecting higher segmentation.
- **Patch density:** Number of patches per unit area.
- **Proximity index (PROX):** A metric that assesses the degree of patch isolation by considering the size and distance of neighbouring patches within a defined search window. $PROX = \sum_{j=1}^{n} \frac{A_j}{D_{ij}^2}$ where *Aj* is the area of neighbouring patch *j*, D_{ij} is the distance between focal patch *i* and patch *j*, and *n* is the number of patches within the search window.
- **Patch richness:** Number of different patch types (classes) present within the boundary of the habitat.
- **Shannon's diversity index:** A relative index to measure patch diversity, based on both the number of patch types (richness) and the distribution of area among them. It is useful for comparing the same landscape over time (McGarigal & Marks, 1995).

All these metrics can be accurately calculated using software as FRAGSTATS (Aranda et al., 2022; McGarigal & Marks, 1995). Additionally, R packages, such as landscape metrics (Hesselbarth et al., 2019), provide an efficient and flexible way to compute landscape metrics while maintaining compatibility with FRAGSTATS outputs.

5 Next steps to address future needs

As environmental challenges continue to intensify across Europe, the establishment of a standardised reporting system for habitat status is becoming increasingly critical. To support effective assessment and management at the European Union (EU) level, it is essential that Member States adopt a common procedure for habitat assessments.

This shared approach will facilitate the integration of diverse datasets, enabling comprehensive analyses and meaningful comparisons across regions. By harmonising terminology, variables, and methodologies, we can ensure that habitat data is both comparable and actionable.

Such standardisation not only improves the quality and coherence of environmental reporting but also fosters collaboration among Member States. Ultimately, it supports the development of more informed, coordinated, and effective conservation strategies that address the urgent needs of Europe's ecosystems.

The harmonisation of procedures for habitat assessment still faces several significant challenges, including:

1. **Establishing common variables**: A shared set of variables must be defined, incorporating both essential and specific variables. This will support the unification of databases across Member States while also allowing for the characterisation of habitat differences within individual national contexts. Standardising these variables will ensure a more coherent data structure that accurately reflects habitat diversity.
2. **Defining methodologies**: It is essential to agree on common or, at a minimum, comparable methodologies. Such alignment will enable reliable data comparisons across Member States and improve the overall quality of assessments. A shared methodological framework will provide clear guidance for data collection and analysis, ensuring all parties work toward consistent standards.
3. **Implementing intercalibration exercises**: Intercalibration is crucial for verifying the accuracy and consistency of methodologies applied across countries. These exercises help identify discrepancies in data interpretation and application, allowing for adjustments that improve overall reliability and validity.
4. **Establishing a standardised reporting format**: A common format for reporting habitat status is needed to enable integration at the EU level. Harmonisation of variables, methods, and units of measurement is essential to streamline data sharing and facilitate effective communication among Member States.
5. **Capacity building for technical staff**: Ensuring the accurate collection and reporting of data requires training programmes for technicians. Building this capacity will equip staff with the skills needed to apply standardised methods effectively and consistently, thereby improving data quality and compliance.
6. **Improving access to technology**: Expanding access to medium-scale technologies – such as drones – for environmental managers is vital for efficient habitat monitoring. In addition, training in remote sensing techniques will enable managers to use these tools effectively, improving data acquisition and analysis.

By addressing these key areas, the EU can move toward a cohesive, standardised, and high-quality approach to habitat status assessment. This, in turn, will support more informed decision-making and enhance the long-term management of natural resources.

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